Assessment of IOP/ISOP Impacts on Freshwater Communities

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1. Introduction

1.1 Rationale for employing aquatic organisms as indicators of environmental change

Environmental impact assessment in the southern Everglades has focused on two classes of environmental factors under the control of water managers: hydropattern and water quality. Both factors can act as ecological "drivers" of aquatic communities in Everglades wetlands, and the impacts of variation in each are well established. Hydropattern is a broad term encompassing all aspects of water delivery, both through volume, spatial distribution, and timing of releases that influence local habitats as water depth and its annual fluctuation, and as hydroperiod, time between marsh-surface drying, and rate of flow (Sklar et al. 2002). Water quality is typically used in a more limited way in the Everglades, focusing primarily on the availability of phosphorus (P) as the key element limiting plant growth (McCormick et al. 2002). While other elements and contaminants (notably mercury) are of known importance in the Everglades, we will not address them in this report. Though hotly debated, mercury appears to be best regulated in the Everglades by diminishing its aerial deposition (Atkeson and Axelrod 2004) and is, thus, outside the realm of Interim Operational Plan (IOP). Also, recent trends systemwide shown decreases in fish-tissue levels of this toxicant.

By their nature, ecosystem impacts of hydropattern manipulation and water quality require different approaches to impact assessment. The Everglades is an oligotrophic wetland ecosystem and all aspects of its geology, paleoecology, and biogeochemistry point toward low P availability since its origins about 5,000 years ago (McCormick et al. 2002; Noe et al. 2001). Anthropogenic impacts of phosphorus can be traced to inflows from canals, through which waters flowing off of agricultural lands are funneled into the Everglades. While there has been controversy about the exact level of phosphorus inflows that adversely affect Everglades aquatic communities, there is a general agreement that a threshold standard for water quality should be very low. Recently, Florida's Environmental Regulatory Commission, based on advice from the Florida Department of Environmental Regulation, set a state Class III water quality standard for water flowing into the Everglades that included an upper bound of 10ppb under many circumstances (ambient water column P levels in the Shark Slough are commonly between 2 and 8 ppb; see Water Quality section of this report). Research conducted in Everglades National Park (ENP) indicated that wetlands there are particularly sensitive to nutrient inflows and that their impacts are cumulative (Box 1). This cumulative impact (termed loading) implies that low levels of nutrient addition continued over time will have the same long-term effects as inflows at higher levels over shorter times. Thus, changes in water delivery that create sustained inflows of even low levels of nutrients into the ENP are not consistent with legal mandates for its management (e.g., 1916 Organic Act; 1964 Wilderness Act).

Box 1. Research demonstrating cumulative impacts of P in Everglades aquatic communities

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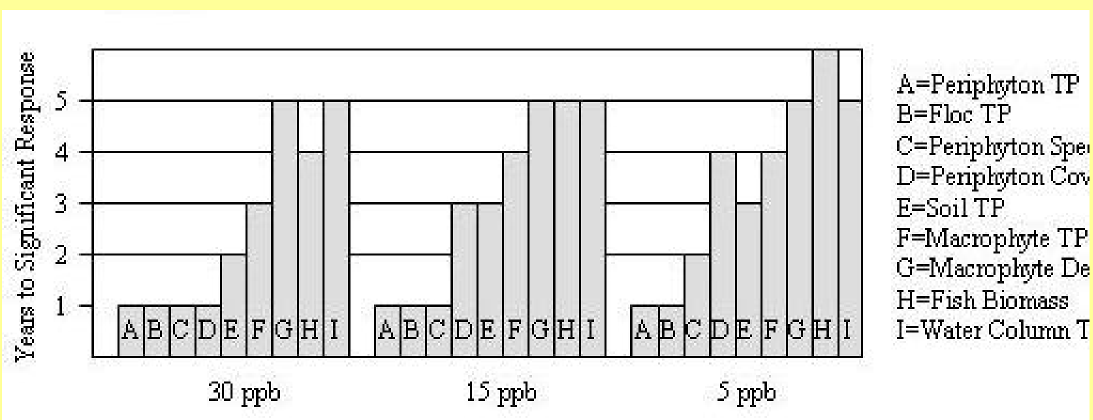
Through funding from Everglades National Park (ENP), the South Florida Water Management District, and the Florida Department of Environmental Protection, we initiated a large-scale P enrichment experiment in 1997. Three 100-m long flow-through flumes were established in Shark River Slough, Everglades National Park (Fig. B1, 1). Each flume contained four channels that received 0, 5, 15 and 30 ppb of P above ambient concentrations. Concentration was kept constant at the head of the channel by continuous calibration with water depth, and downstream delivery was dependent on natural water flow, which was parallel to the orientation of the flume channels (Childers et al. 2001, Noe et al. 2002). After 5 years of continuous dosing, significant responses were detected in all ecosystem parameters at all dose concentrations (Fig. B1.2). The marsh responded dynamically to dose, with effects first becoming visible in the microbial community, followed by the sediments, macrophytes and consumers. Water column P concentration was the last measured variable to show a significant response, indicating that downstream transport of P was primarily through biota rather than the water column (Gaiser et al. 2004). By the end of the 5th year, the most conspicuous differences between treated and control channels were the decreased biomass of floating calcareous periphyton mat and increased densities of the dominant emergent macrophytes in all dose channels. All biotic changes preceded detectable increase in water column P, indicating rapid biotic uptake by periphyton, floc, sediments and plants. A compendium report is available at http://serc.fiu.edu/periphyton/.

Fig B1.1. Aerial photo of one of the flumes used for this study.

Figure B.1. 2. Time (years) to significant change for 8 categories of responses at locations 15-m downstream of 5, 15 and 30 ppb P inputs. The sequence of responses is similar among dose-levels but the rate of progression of significant responses is dose-dependent.



In contrast to water quality, targets for hydrological management are more difficult to derive for several reasons:

1. Everglades hydrology is highly variable in response to seasonal and inter-annual variation in rainfall;
2. Hydrological parameters such as hydroperiod, annual minimum water depth, and annual maximum water depth are spatially heterogeneous and their control is complex and incompletely understood (see Hydrology section);
3. Soil subsidence, diminution of ecosystem size, and sea-level rise contribute to a list of factors complicating hind-casting historical hydrology and diminishing the options of managers to re-create historical hydrological patterns.

Thus, it is difficult to identify “desirable” and “undesirable” hydrological targets for specific locations. In spite of these challenges, it is clear that the Shark Slough (especially Northeast Shark Slough), the Rocky Glades, and Taylor Slough experienced much longer hydroperiods historically and as recently as the 1920’s and 30’s. The goal of lengthening hydroperiods in these areas, as part of ecosystem “restoration”, is supported by historical reports (e.g., Willoughby 1898), observational accounts (e.g., Simmons and Ogden 1998), and computer simulations (Fennema et al. 1994). Thus, in the Comprehensive Everglades Restoration (CERP) managers have incorporated the extension of hydroperiods in these areas as a major goal for restoration of the Everglades (USACOE & SFWMD 1999). The dynamic nature of water tables in wetlands has rendered assessment of hydrological management by traditional multimetrics challenging (Wilcox et al. 2002). Thus, our goals for biotic-hydrological assessments in this report are to elucidate the impacts of changes in hydrological operations on selected aquatic communities that, collectively, provide insight to food-web structure and function (Box 2) in the areas affected by IOP/ISOP.

1.1 Box 2. Food-web Definitions

Food-web structure is a description of the diversity and relative abundance of organisms in a location as it relates to their feeding inter-relationships. In this report, it is used similarly to community structure.

Food-web function is a description of the relative movement of nutrients or energy through a food web to support key species (for example, food for wading birds) or ecosystem attributes (for example, water purification). Though difficult to assess directly, it is an emergent property of food-web structure. In this report, we treat it as an outcome linked to the abundance of species targeted for monitoring as critical feeding links to masthead species (e.g., alligators, wading birds in general) and threatened species (e.g., Wood Storks).

Ecological impact assessment in the Everglades has focused on the identification of Performance Measures that are sensitive to hydrological and nutrient impacts (Ogden et al. 2003). Ideally, biological Performance Measures incorporate inherent targets indicative of successful management. While targets are relatively clear-cut for monitoring nutrient impacts, targets for hydrological impacts are more challenging. In the case of ecological assessment for IOP, we employed two types of biological Performance Measures for nutrients (periphyton Total Phosphorous (TP) and midge pupae relative abundance), and three for hydrology (floating mat volume, macroinvertebate and fish metrics). Our targets for nutrients were to identify all new phosphorus enrichment above ambient that resulted from operational changes under IOP/ISOP. Our targets for hydrology were to identify cases where hydrological conditions had not progressed toward the goal of wetter conditions in Shark Slough, Northeast Shark Slough, the Rocky Glades, and Taylor Slough. Hydrological targets for Water Conservation Areas (WCA)-3A and WCA-3B were less clear than for ENP, but in most cases shorter hydroperiods were not desirable. The exception was in southeastern WCA-3A where ponding was also considered undesirable.

Range expansion by non-native taxa, particularly new invasions of the ENP, is a third area of concern for ecological impact assessment in CERP (Ogden et al. 2003). Operations and structural changes that facilitate the introduction of non-native taxa into the National Park are clearly inconsistent with the legal mandates governing its management. In this report, we have noted changes in species composition of fishes and aquatic macroinvertebrates as part of our ongoing monitoring in ENP, particularly in the areas where new connections to canals were created.

We report on assessments of seven biotic groups from five regions of the Everglades in our evaluation of the impact of IOP/ISOP (Table 1). Each biotic group included multiple taxa or parameters that were assessed as performance measures. For example, periphyton tissue TP was assessed in transects originating at the inflow points of the S-332B, C, and D detention areas into the Taylor Slough headwaters of ENP. The target for this performance measure was values not to exceed 200 ug/g (Gaiser et al. 2004). Other performance measures and their targets are discussed in the sections that follow. Those performance measures are linked through their roll in the food web in providing sustenance to top carnivores such as wading birds and alligators (Figure 1). Both hydrological variation and nutrient enrichment fit into this food-web perspective by altering the relative abundance and availability of key food items and, thus, routes of energy to these top consumers that are important to managers and the public. Collectively, our biotic monitoring framework can be viewed as tracking the environmental impacts of management on key indicators of food-web support of critical species and ecosystem function in the areas affected by IOP/ISOP.

Table 1. Summary of the regions and biotic groups monitored to identify nutrient and hydrological impacts from IOP/ISOP. The type of data evaluated for this report is listed. * Small fish were monitored for evidence of non-native taxa in S-332, not nutrients.

	Regions				
	Nutrients	Hydrology			
Biotic Group	S-332	Rocky Glades	Taylor Slough	Shark River Slough	WCA-3A&B
Periphyton	Tissue TP		volume	volume	volume
Midge larvae	Relative abundance of sensitive and tolerant taxa	Relative abundance of sensitive and tolerant taxa	Relative abundance of sensitive and tolerant taxa		
Small crustaceans (copepods)		Species composition			
Grass shrimp			density	density	density
crayfish			Relative abundance of species	Relative abundance of species	
Small fish	Non-native taxa* (Minnow traps)	CPUE and non-native taxa (Minnow traps and drift fences)	Density (Throw trap)	Density (Throw trap)	Density (Throw trap)
Large fish			CPUE (boat electrofisher)	CPUE (boat electrofisher)	CPUE (boat electrofisher)

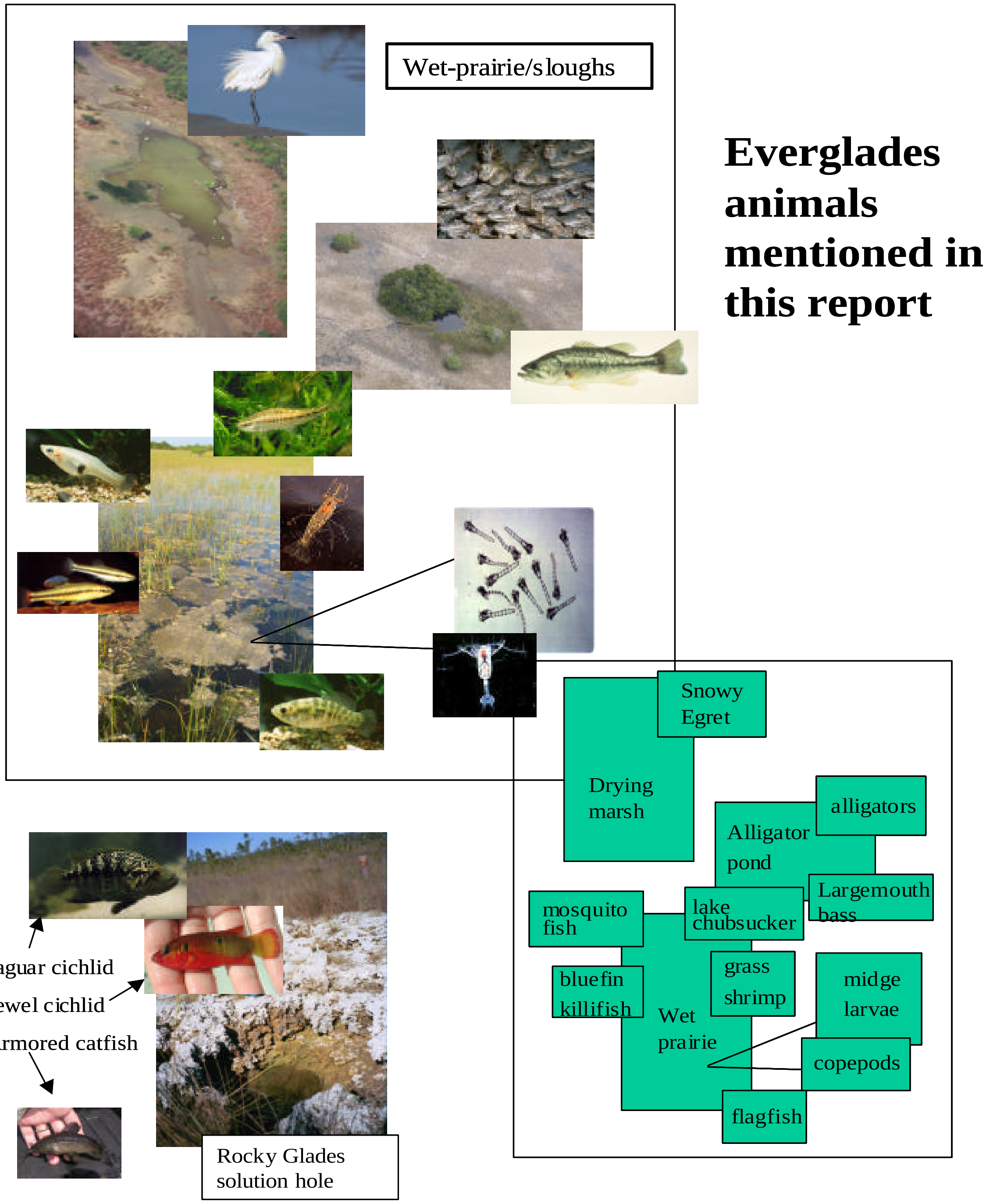


Figure 1. Pictures of selected animals discussed in this report. Note that wading birds and alligators are "masthead" species that were not monitored for IOP/ISOP impacts because they are not indicators of impact over the timeframe of IOP/ISOP. Most of the monitoring discussed was conducted in wet-prairie sloughs or Rocky Glades habitats. Wet-prairie animals are especially susceptible to being consumed in dry seasons when marshes dry. Large fishes are primarily confined to alligator ponds during the dry season. Midges and copepods primarily inhabit floating mats of periphyton.

1.2 Linking operations to biological parameters: Data analysis and interpretation

When possible, we employed a Before-After-Control-Impact (BACI) statistical analysis of our data to test for significant differences between the pre- and post-IOP/ISOP time period (pre-IOP/ISOP defined here as 1996 – 1999 and post-IOP/ISOP as 2000 – 2002). We have information that can be assessed in this way for density of small fishes, catch-per-unit-effort (CPUE) of large fishes, floating mat volume, emergent vascular-plant stem density, and freshwater prawn density. These data are from 22 study sites in Shark Slough, Taylor Slough, and WCA-3A and WCA-3B. We have been sampling from three plots at each site several times per year since 1996. Our model for testing the BACI hypothesis treated samples collected at the same plot as a repeated measure that was modeled with a one-step autoregressive model to account for temporal autocorrelation (Murtaugh 2002). We tested the significance of the site-by-treatment interaction (where treatment was before and after implementation of IOP/ISOP) with the use of an analysis of deviance (ANODE). Our application of ANODE entailed comparing Akaike's Information Criterion (AIC) for a statistical model lacking the site-by-treatment interaction with an identical model including it (Burnham and Anderson 2002). When the difference of the two AIC's was equal to or exceeded two, the interaction added significant information to explaining the data. This approach is particularly desirable for fitting complex mixed models, as is the case in this study (Burnham and Anderson 2002; Littell 2002).

For impact assessment of nutrients, we report an analysis of the relative abundance of environmentally sensitive and tolerant midge larvae to evaluate nutrient and hydrological impacts associated with the new S-332 structures at the eastern headwaters of Taylor Slough. This approach has proven particularly useful for nutrient assessment in areas where a time series of data has yet to be obtained. Background research to identify midge taxa useful for impact assessment is highlighted in Box 3. Midge species are often very limited in their habitat requirements (the reason they are good indicators) and current work suggests that different taxa are preferred indicators in the northern Everglades (WCA-2A) than in ENP. While research toward selecting optimal midge indicators for the Park is ongoing, we report responses of indicator groups developed by our research in ENP, as well as other published information on midge water-quality relationships (Adamus and Brandt 1990; Epler 2001; King 2001; Adamus and Danielson 2001). Though not free from the concerns of experimental design motivating the use of BACI, the preliminary work in selecting midge indicator taxa provides some support for the assertion that their presence is solely indicative of nutrient enrichment. Though limited, we do report some pre-IOP/ISOP data of this type from key study sites to help interpretation. Ultimately, we will need to monitor these indicators to assess future dynamics that would be indicative of environmental impacts, should they be present (i.e., our current assessments of nutrient impacts of new IOP/ISOP structures and operations must be viewed as cautionary).

Box 3. Development of midge Indicator-Species Groups for assessing nutrient enrichment in the Everglades

R. Jacobsen

Our water-quality assessments using midge community composition are based primarily upon changes in the relative proportions of groups of midge species that have been found to either increase or decrease in relative abundance in response to phosphorus enrichment. This reliance on analysis of indicator species is somewhat of a departure from the use of community attributes commonly used in stream bioassessment such as taxonomic richness, taxonomic structure, and feeding ecology. However, King (2001) found that metrics based upon community attributes were ineffective for Everglades assessment because they failed to show monotonic relationships with nutrient enrichment. Our indicator species groups are drawn from two independent research studies examining invertebrate community change along canal discharge plumes into the Everglades: (1) King's dissertation research on determinants of invertebrate community composition along the strong nutrient gradient in WCA-2A (King 2001), and (2) work on midge-community changes near canal inflows into Everglades National Park (Jacobsen and Perry 2002). We tested the reliability of nutrient-sensitive and nutrient-tolerant indicator species groups derived from both gradient studies by examining their responsiveness to low levels of experimental P dosing in the Shark River Slough flumes (Box 1). Midge-pupal exuviae were collected in March 1999, approximately 5 months after dosing started, and again in January 2001.

Water column total P did not change within the flume channels during this study. However, significant changes were eventually observed in every biotic community component examined, thereby demonstrating the importance of using bioassessment techniques for detecting nutrient enrichment in the Everglades. ENP-sensitive taxa (species that decrease in abundance near canal inflows into ENP) were more responsive to nutrient enrichment (abundance declined with increasing levels of P-dosing) than WCA-sensitive midge species (Fig. 1). However, taxa identified as tolerant of enrichment in WCA-2A, responded much more positively to P-dosing than ENP-tolerant taxa. The greater responsiveness of ENP-sensitive species and WCA-tolerant species probably reflects differences in the degree of enrichment and steepness of the nutrient gradients sampled in the process of selecting those species as indicators. Several midge species, such as *Dasyhelea* cf. cincta and *Dicrotendipes* sp. A Epler, were not found to be indicator species in nutrient-gradient studies, but were highly responsive to P-loading in Shark River Slough flume experiments (Fig. 2).

Box 2. Continued.

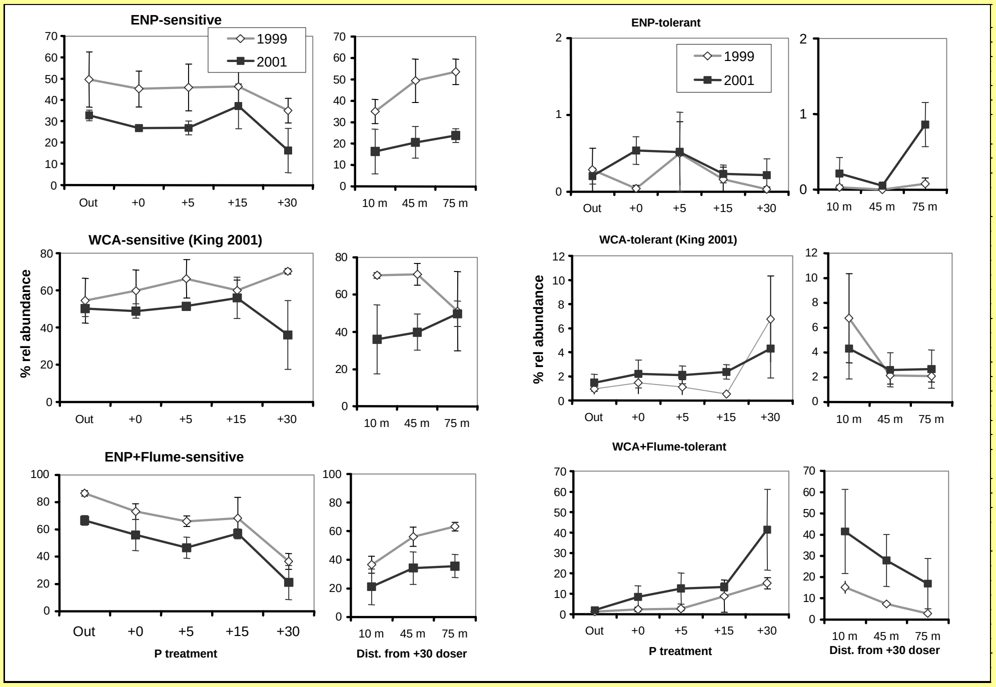


Figure 1. Responsiveness of water quality indicator groups derived from gradient studies in Everglades National Park (Jacobsen and Perry 2002) and Water Conservation Area 2A (King 2001), to P-dosing in the 3 Shark River Slough flume arrays. The taxonomic composition of each indicator group are listed in Table 1.

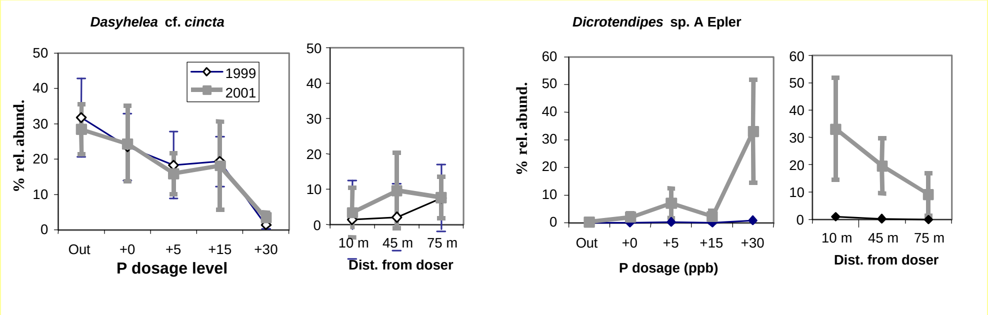


Figure 2. Response of *Dasyhelea* cf. *cinct*a and *Dicrotendipes* sp. A Epler to P-dosing in Shark River Slough flume arrays, 1999 and 2001.

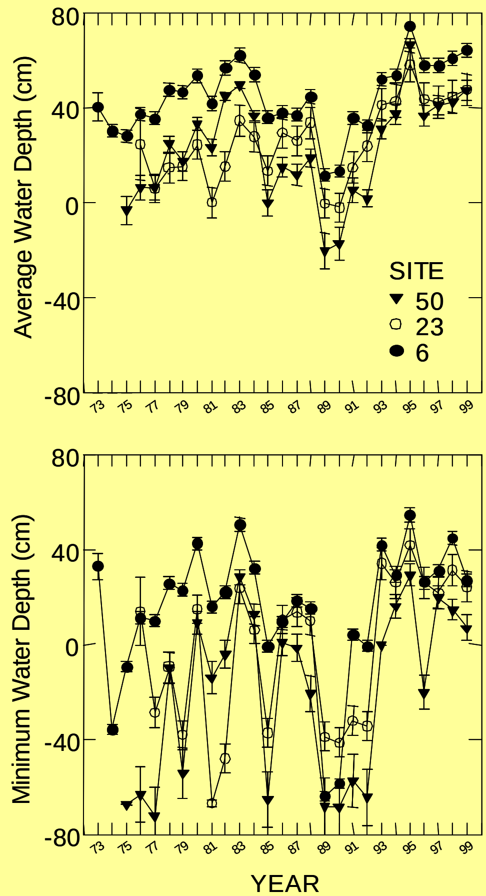
1.3 Targets for Assessment of Hydrological Management

The relationship between rainfall and water depth at our study sites is complex, characterized by time-lagged relationships with rainfall at upstream parts of the ecosystem that is mediated by structural operations (Box 4). The displacement of upstream rainfall to downstream habitats (e.g., rainfall over WCA-3A to Shark Slough) is the crux of water management for ENP. Hydrological BACI analyses (reported in the Hydrology section of this report) indicated that the IOP/ISOP period had lower dry-season hydrological stages and shorter hydroperiods than the pre-IOP/ISOP period at many sites in Shark Slough, Taylor Slough, and in WCA-3A and WCA-3B. Those impacts were felt strongly in WCA-3A, where minimum water depths were last exceeded in the drought years of 1989-1990. In June 1989, all of southeastern WCA-3A dried (Bancroft et al. 1994: 631), an area with the longest hydroperiod of the southern Everglades because of ponding caused by the Tamiami canal and L-67 levee. While this area did not completely dry in 2000, much of WCA-3A and WCA-3B did. This was not expected under pre-IOP/ISOP management in a year with low, but not unusually low (a one in four year), rainfall south of Lake Okeechobee. In this case, the impact of a very dry year north of Lake Okeechobee (Smith et al. 2003) was transmitted into the southern Everglades by management of canal stages. We are not aware of methods to assess the impact a drought north of Lake Okeechobee on the southern Everglades prior to human intervention, in a year with moderate rainfall in the southern region. However, it is likely that the deep peat soils typical throughout the northern Everglades would have held water and buffered such an effect to a greater extent than is seen today. Further, socially driven diversion of water in Lake Okeechobee and elsewhere did strongly influence canal-stage management in 2000.

Since the hydrological BACI analyses indicated that all of our long-term study sites (see map, Figure 2) experienced lower dry-season water levels than expected under IOP/ISOP (dry season 30-Day moving averages), there were no “control” areas for us to perform a standard BACI analysis (see Hydrology section for details and maps). However, we could assess overall impacts by ranking the relative impact expected based on the estimated magnitude of deviation from pre-IOP/ISOP reported in the Hydrology section. First, we expected that ecological effects linked to marsh drying in WCA-3A and WCA-3B in 2000 would not have happened under pre-IOP/ISOP operations (a one-in-four dry season would not have caused widespread drying). Second, 30-day moving-average depths suggested that dry-season IOP/ISOP impacts might be revealed by greater differences between pre- and post-IOP/ISOP periods at both WCA-3A, WCA-3B, and Shark Slough study sites, than in Taylor Slough (with the possible exception of sites in Craighead Pond, location of water gauge P-37). Hydrological analyses suggested that marsh dry-downs might have occurred at NP-202 and NP-203, but not at P36 or P37, under pre-IOP/ISOP operations. Thus, we targeted study sites in southern Shark River Slough near Rookery Branch and Taylor Slough in Craighead Pond for IOP/ISOP effects. In a broader sense, CERP goals are to lengthen the hydroperiod in Shark and Taylor Sloughs, so drying in those areas in years with only modestly less rainfall than average was not consistent with long-term goals for management of ecosystem restoration and management of ENP.

Box 4 .nalysis of the role of upstream flow through control structures in determining water depth in northern Shark River Slough.

Figure B1. Hydrographic parameters from the three study sites from 1973 to 1999. A. Average water depth was estimated as the mean of monthly averages at each plot. B. Minimum water depth was the annual minimum at each plot. Means with one standard error interval bars are reported in both graphs based on inter-plot variance, N = 3 plots at each site.



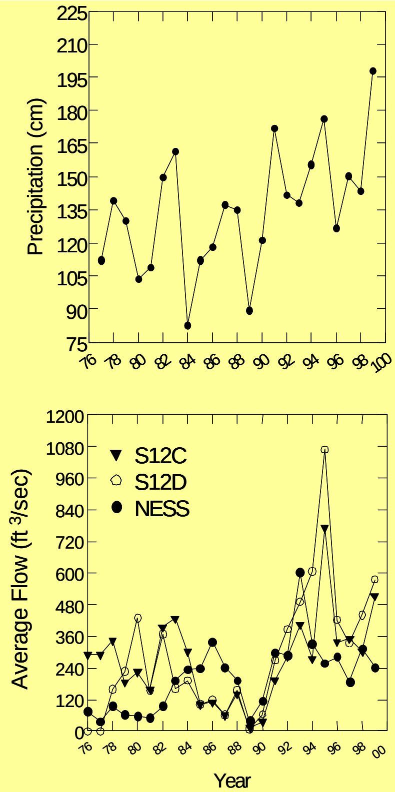
To illustrate the relationship of rainfall and operations on local water depth in Shark River Slough, we analyzed of the relationship between water depths (Fig. B1) at three study sites in northern Shark River Slough (map in Fig. 1: SRS50, SRS6, SRS23) and flow rates from the S-12 structures in Shark River Slough, culverts east of L-67-E into Northeast Shark River Slough (Fig. B2). Precipitation records were obtained from the Tamiami Ranger Station (formerly Forty-Mile Bend), the closest rain gauge to our study sites that was continuously monitored between 1977 and 1999. Flow rates were obtained from the South Florida Water Management District and U.S. Geological Service. We used backwards stepwise multiple regression to assess the relative contribution of each water source to water depth at the study sites.

Results: Control structures S-12A, B, C, and D permit water to enter the Shark River Slough upstream of sites 6 and 50, while water enters Northeast Shark River Slough through a series of about 20 culverts. Average monthly water depths at Sites 6 and 50 could be predicted best by water flow through structures S-12C and S-12D, while regional rainfall yielded little indication of an influence (Multiple regression results, **Site 6**: Precipitation $t_{1,263} = 0.796$, $P = 0.427$; S-12A $t_{1,263} = -0.732$, $P = 0.465$; S-12B $t_{1,263} = -0.940$, $P = 0.348$; S-12C $t_{1,263} = 6.473$, $P < 0.001$; S-12D $t_{1,263} = 4.630$, $P < 0.001$; $R^2 = 0.525$; **Site 50**: Precipitation $t_{1,263} = -0.257$, $P = 0.797$; S-12A $t_{1,263} = 0.014$, $P = 0.989$; S-12B $t_{1,263} = -1.323$, $P = 0.187$; S-12C $t_{1,263} = 6.919$, $P < 0.001$; S-12D $t_{1,263} = 4.227$, $P < 0.001$; $R^2 = 0.559$). Average monthly water depth at Site 23 was influenced by regional rainfall, water flow

Box 4. continued

through the NESS culverts, and also flow through structures S-12C and S-12D (Multiple regression results, **Site 23**: Precipitation $t_{1,262}$ = 2.620, P = 0.009; S-12A $t_{1,262}$ = -0.455, P = 0.650; S-12B $t_{1,262}$ = -0.944, P = 0.346; S-12C $t_{1,262}$ = 5.092, P < 0.001; S-12D $t_{1,262}$ = 3.562, P < 0.001; NESS $t_{1,262}$ = 8.541, P < 0.001; R^2 = 0.546). Rainfall affected water level at the study sites, but primarily through its influence on flows from the north. Low rainfall years (annual total < 1m) occurred in 1984 and 1989 (Fig. B1.A), and water flow into the Park was low from all sources in 1989 and 1990 (Fig. B1.B). Rainfall and water flow into the Park were relatively high (for this study period) from 1992 through 1999 (Fig. B1). Water flow into Northeast Shark River Slough, relative to the sum of flows through the S-12 structures into Northwest Shark River Slough, increased after 1983.

Figure B2. Regional patterns of precipitation and water flow into the study areas between 1977 and 1999. A. Annual rainfall (summation of daily values) collected at the Tamiami Ranger Station (aka, Forty-Mile Bend) gauging station. B. Average flow of water into Everglades National Park through selected water structures. These sites were chosen because of their influence on the study



Discussion: The environmental factors controlling water depth at our three study sites differed between the northwestern and northeastern sections of Shark River Slough. Water depth at sites SRS6 and SRS50 was correlated with flow from water delivery structures S-12C and S-12D. While it would be desirable to have precipitation data from gauges closer to the sites, we found no evidence that such information would alter our finding that precipitation affected these sites largely or solely through its influence on management of flow through the water control structures. Northeast Shark River Slough differed in this regard, where our measure of regional rainfall did improve our regression model of water-depth fluctuation. That analysis also indicated management actions that changed flows through the culverts upstream of Site 23 affected water fluctuations there. The additional influence of water deliveries from S-12C and S-12D must have been through groundwater or from back-flow around the L-67E levee. The combined flow through the S-12 structures greatly exceeded the combined flow through the culverts for most of the study, with the possible exception of 1985 through 1987. Flow into Northeast Shark River Slough was increased after 1984 as a result of the Experimental Water Delivery 1, and its primary effect in our data was manifested as increasing the annual minimum water depths at Site 23.

2. Nutrient Impacts in Taylor Slough Headwaters & C-111/panhandle

2.1 Periphyton

We examined TP in periphyton samples from the vicinity of S-332B, S-332C, and the inflow of water into ENP at the old S-332 structure (downstream of S-332D) as a sensitive test for nutrient runoff into these areas. Periphyton accumulates environmental phosphorus, and levels exceeding 200 ug/g indicate potential anthropogenic enrichment in the Everglades (Gaiser et al., 2004). Sampling was conducted on November 10, 2003, following standard methods. When possible, three 6-cm^2 cores were taken from floating periphyton mats and analyzed separately for TP, TN, and % inorganic carbon. Those samples were taken after a period of unusually high water resulting from extended rainfall.

Figure 2. Aerial photo of S-332B showing the direction of the transect (red arrow) where periphyton was sampled. The red arrow is approximately 60-m long. The black rectangle indicates the approximate area where nutrient enrichment was noted.

We observed elevated phosphorus in the immediate vicinity of the berm surrounding the S-332B structure (Figure 2). Periphyton samples in the vicinity of the S-332B berm were visibly different from those taken further into the marsh (darker green, more filamentous). Laboratory analysis indicated that these samples held elevated levels of TP (Figure 3). More typical mats and epiphytic algal growths were noted at 50 m distance from the berm. This transect originated south of the cement sluiceway where water overflowed directly into the marsh in 2002.

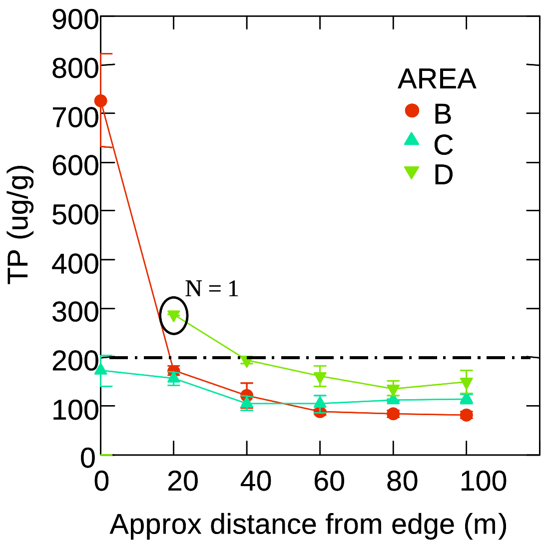


Figure 3. Total phosphorus from periphyton samples taken along transects from each S-332 impoundment or inflow point. Only the 0 to 20m distances at S-332B are different (P<0.05). The dashed line indicates a threshold for enrichment.

There was no nutrient gradient observed in the periphyton transect samples taken from S-332C. There is no direct water flow from this detention area; where all water release was by seepage.

We also observed evidence of elevated TP levels approximately 20 m from the point of water inflow into ENP at the old S-332 structure. The emergent vegetation (spikerush, willows, cattails) was very dense and tall in the first 60 m from the inflow point. There was very little periphyton in this area, where water depths were approximately 46 cm at the time of our visit. We found no periphyton to sample at the canal inflow point, and only one sample could be accumulated at 20 m from the inflow. This sample indicated a relatively high level of TP, but could not be shown to differ from the more distant samples because of the lack of replication (Figure 3). The data must be interpreted with caution because relatively low-nutrient water flowing into an oligotrophic Everglades wetland, as is the case in this area, can lead to elevated nutrient levels from particulate re-suspension and local loading.

We caution against strong statements regarding nutrient effects of the new impoundments based on this one-time transect sampling. The nutrient patterns observed could be derived from an initial release of nutrients from the impoundments accumulated when they were in agriculture, but that will be purged in a relatively short time with little ill effect. However, our data raise the potential of nutrient release via groundwater seepage from S-332B because the lone overflow event occurred more than a year before the samples were collected and periphyton integrate water-column nutrients over a relatively short time period. Continued monitoring and assessment is needed.

2.2 Midges in Taylor Slough and its headwaters, and in C-111/ Eastern panhandle

Midges (Diptera: Chironomidae & Ceratopogonidae) are widely recognized as indicators of organic enrichment in lentic environments (Rosenberg 1992; Lindegaard 1995), and they are increasingly used to assess other pollutants (Williams et al. 2001; Wright et al. 1996; Ruse 1985). Because of their high species richness, the great diversity of microhabitats that they occupy, and their wide collective tolerances for physical and chemical conditions, studying midges alone can be as effective as using a larger group of invertebrate taxa in bioassessment (Armitage & Blackburn 1985; Ruse 2002; Wright et al. 1996). In the northern Everglades, King & Richardson (2002) found that chironomid midges, when identified to species, were the most informative group for detecting nutrient enrichment in WCA-2A (see Box 3 for a study from ENP).

The midge-monitoring program in ENP has focused on using the composition and dynamics of midge communities as a tool for ecological assessment. We sample midge communities primarily by collecting their floating pupal exuviae, according to U.S. Environmental Protection Agency (USEPA) protocols. This method of sampling allows us to collect and process larger numbers of midge specimens with much higher efficiency than is possible through conventional substrate sampling for larvae and pupae. This greater efficiency enables a more comprehensive description of midge-community composition for making comparisons among different sampling sites.

2.2.1 Taylor Slough

In October 2001, three replicate quantitative pupal exuviae samples were collected from spikerush and sawgrass-dominated habitats at four locations along Taylor Slough: 50 meters southwest of S332, just west of the SR 9336 bridge (about 2 km south of S332), approximately 5 km south of S332, and approximately 10 km south of S332.

Comparisons of the proportions of groups of sensitive species at each site suggested that water quality was highest at the two sites farthest from S332, with sites approximately 3 km south of State Road (SR) 9336 supporting midge assemblages indicative of the best water quality. Proportions of all sensitive species groups were lowest near the L-31W canal. Tolerant species groups incorporating ENP-tolerant species were significantly greater near the L-31W canal and just west of the FL SR 9336 bridge than at sites downstream of Royal Palm. Adjacent to the L-31W canal, elevated levels of ENP-tolerant taxa result from a number of those constituents. However, only one “tolerant” species was present near SR 9336, and this reduction in numbers of species tolerant of nutrient enrichment suggests that water quality near the SR 9336 bridge may actually be better than indicated in Figure 4.

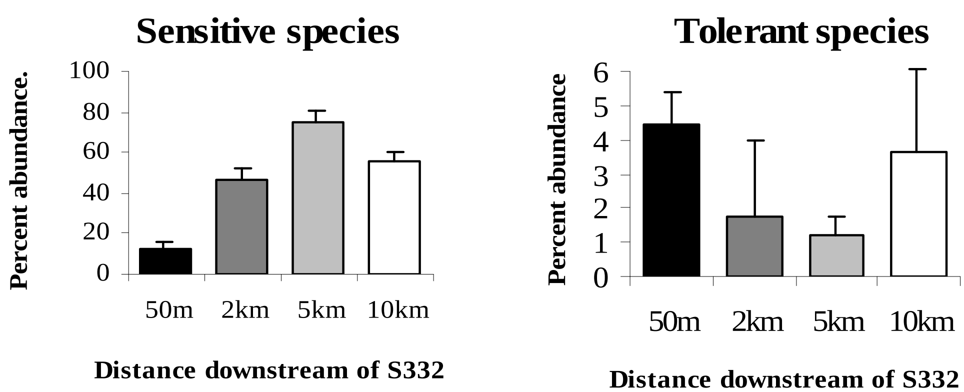


Figure 4. Percent relative abundance of nutrient-sensitive and nutrient tolerant species groups at Taylor Slough marsh sites located 50m, 2 kkm, and 10 km downstream from S-332 on L-31W canal.

Our sampling near the S-332 pump station in 1998-1999 allowed an opportunity to compare midge communities at two sites before and after pump operation was halted in February 2000. However, we can only compare single samples collected from the same site at the same time of year. Site NTS-3 is located about 50 meters north of the S332 pump station. NTS-3 failed to support calcareous periphyton, probably because of currents generated by operation of the pumps, so we added NTS-5 to our study. NTS-5 was located 50 meters north of NTS-3, was less affected by pump operation, and supported calcareous periphyton. The site labeled “East 2002” is the same site as site 332D-2, and is on the east side of the L-31W canal opposite NTS-3. In October 1999, midge communities at NTS-3 supported far less nutrient-sensitive (nutrient-intolerant) species than NTS-5. In October 2002, the midge community at NTS-3 had a slight increase in nutrient-sensitive species, but has a large increase in nutrient-tolerant species (Figure 5). This suggests that there has been no improvement in water quality since the pumping at S332 was halted.

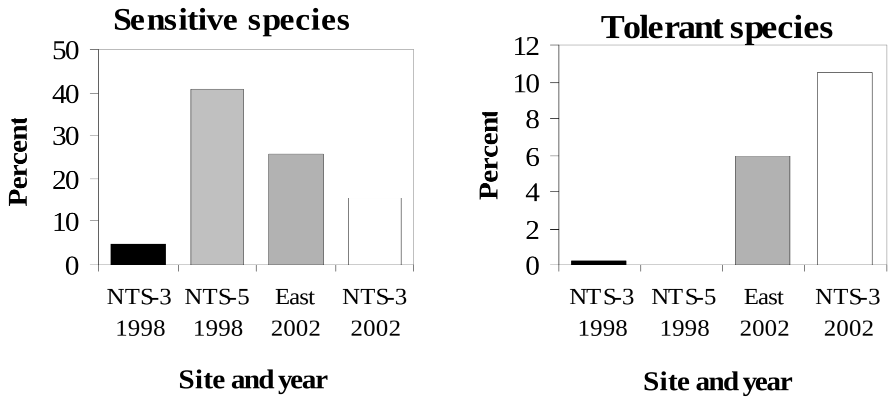


Figure 5. Proportions of nutrient-sensitive and nutrient-tolerant species at sites NTS-3 and NTS-5 in October 1998 (west of L-31W, and NTS-3 and the marsh east of the L-31W canal opposite NTS-3 in October 2002.

Site NTS-2 is located 50 meters south of the S332 pumps. Decreases in nutrient-sensitive species groups, coupled with increases in nutrient-tolerant species groups in 2001, again indicated that conditions may have become more degraded since the pumps were shut off (Figure 6).

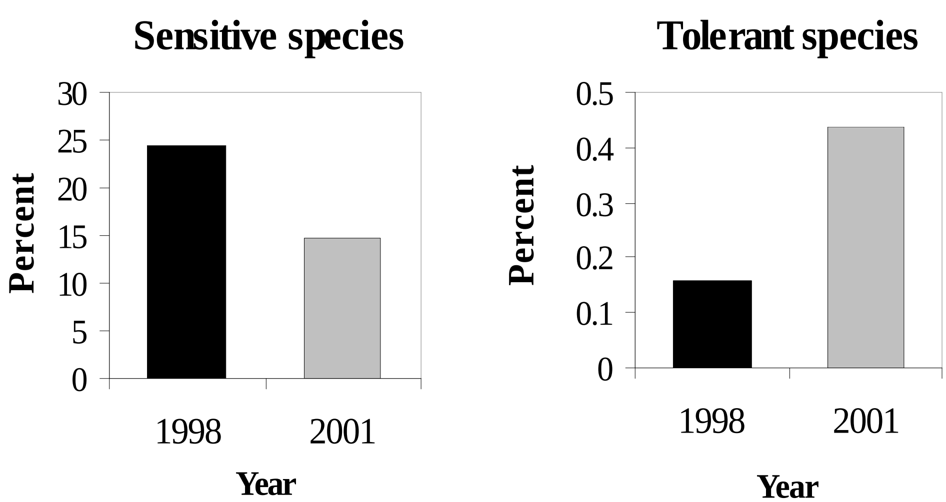


Figure 6. Proportions of nutrient-sensitive and nutrient-tolerant species at upper Taylor Slough site NTS-2 (50 m SW of S-332) in October 1998 and October 2001)

2.2.2 ENP Marshes Near S-332 Detention Areas

On 13-14 November 2001, two quantitative midge-pupal exuviae samples were collected each from sawgrass and *Muhlenbergia* habitats in ENP at approximately 50 meters and 1000 meters respectively west of the S-332B detention area spillway. Water, soil, and plant tissue samples were also collected from each plant habitat at each sampling site. Two additional pairs of quantitative midge samples were collected near the eastern inflow into the retention pond, and near the western spillway. Water levels in the detention area did not exceed the height of the spillway on the days sampled. However, it was apparent that water levels in the detention area had recently exceeded the height of the spillway, and large volumes of water had flowed directly into ENP.

Detention area operations that resulted in direct discharges into ENP appeared to have elevated pupal exuviae abundance and species density in marshlands receiving these waters. Total species richness was very high in the S-332B detention area and in adjacent Rocky Glades marshes relative to more typical Everglades habitats. Fifty species were collected from the two sites in the S-332B detention area (total of four $1m^2$-samples). Sawgrass and *Muhlenbergia* habitats located 50 meters west of the detention area spillway yielded a combined total of 44 taxa (total of four $1m^2$-samples).

Shifts in the proportions of midge indicator groups also suggested that the detention area, and its accidental discharges, affected habitat quality for neighboring midge communities. Proportions of ENP-sensitive species declined, while ENP-tolerant, and to a lesser extent, WCA-tolerant groups increased (Fig. 7). What is most notable about the midge community near S-332B, was the presence of species such as *Chironomus stigmaterus*, *Goeldichironomus holoprasinus*, *Goeldichironomus amazonicus*, *Procladius bellus*, and *Dicrotendipes* modestus, all of which are considered to be indicators of eutrophic conditions in wetlands (Adamus and Brandt 1990).

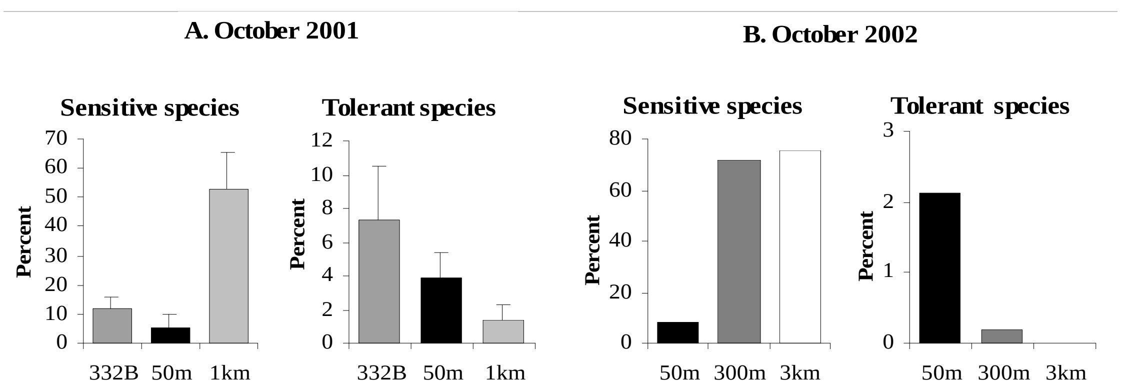


Figure 7. Proportion of nutrient-sensitive and nutrient-tolerant species in midge pupal exuviae samples collected in or near detention area S-332B. A- October 2001 from S-332B detention area, and from sites 50 m, and 1000 m west of the detention area. B.- October 2002 from Cladium sites 50 m, 300 m, and 3 Km west of S-332B detention area

Changes in midge community composition along sample transects from the detention area westward into ENP in October suggested the retention ponds and L-31W affect water quality in ENP (Fig. 8). Percentages of ENP-sensitive and ENP & flume-sensitive species groups were consistently reduced near the western boundary of the detention areas. Groups of species tolerant of nutrient enrichment (both ENP and WCA-2A tolerant species groups) also increased significantly near the detention areas. The lack of enrichment-tolerant species in the retention ponds appeared to be an artifact of the detention areas overall dissimilarity to Everglades marshes. Several of the most abundant species in retention ponds (e.g., *Apedilum elachistum*, *A. subcinctum*, *Polypedilum nubifer*, *Tanypus stellatus*) are widely reported to reach nuisance population sizes in eutrophic waters (Ali 1995) and are considered indicators of nutrient enrichment (Adamus and Brandt 1990). However, those species are rare or absent in both ENP and WCA-2A, and consequently, were not identified as indicators of enrichment in earlier studies (King 2001, Jacobsen and Perry 2002). *Polypedilum nubifer* is a notorious pest species in shallow, eutrophic waters in Asia, Australia, and Hawaii. It had never before been reported in Pan-America before and may be a recent introduction to south Florida waters (Jacobsen and Perry, 2004).

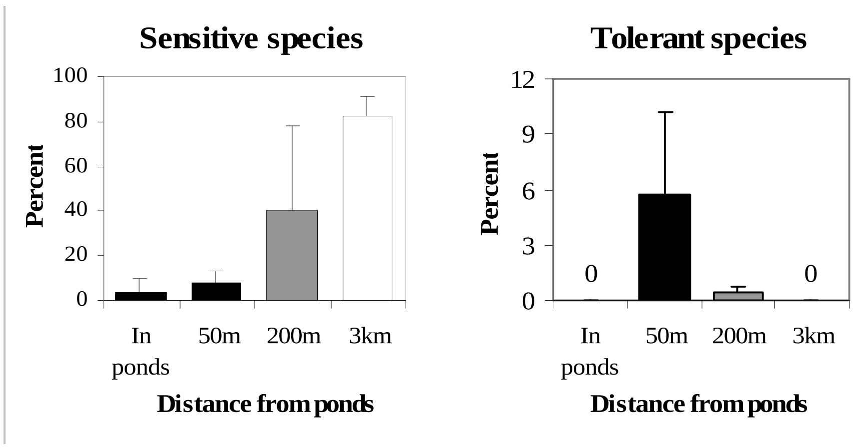


Figure 8. The relative abundances of nutrient-sensitive and nutrient-tolerant midge species groups in October 2002 samples from the detention areas, and from sawgrass-dominated sites 50 m, 200 m, and 3 km west of the detention areas.

We conducted sampling in late December at all sites that had surface water. Everglades marshes near the 332B detention area have numerous solution depressions, and samples were collected from two depressions approximately 50 meters west of the detention area, and an additional sample was collected about 500 meters west of the detention area.

Proportions of midge-indicator species groups also suggested that water quality improved along the western side of the retention pond (Figure 9). Midge community data indicated some degradation near the detention area, even though the western portion supported a relatively healthy assemblage. This may represent a lingering effect of direct flow into the marsh during operation of the detention area in 2001.

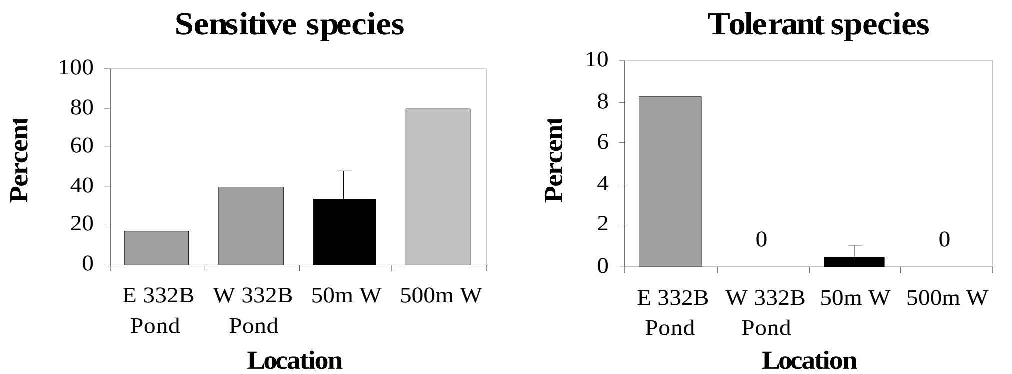


Figure 9. Proportions of midge nutrient indicator groups in pupal exuviae samples collected from the S-332B detention area and from sawgrass-dominated solution depressions 50 m and 500 m west of the detention area, ENP, December 2002.

Midge indicator group percentages suggest that the S-332C detention area was more enriched than the S-332D detention area. Detention area S-332D supported much higher proportions of species considered to be intolerant of enrichment than either S-332C or the two marsh sites. Species tolerant of enrichment were more prevalent in S-332C than in S-332D, particularly WCA-2A tolerant species, which are probably the most reliable tolerant group of species for indicating degradation. Poor-water-quality species, such as *Chironomus stigmaterus*, were collected from the 332C detention area. The site south of S- 332D. 30 meters east of L-31W, is primarily wetted from spillover from the canal. The site 50 meters west of 332C yielded only a small number of midge pupal exuviae in general (21 pupal exuviae), and very few exuviae of indicator species.

2.2.3 Taylor Slough Near The C-111 Canal

In Taylor Slough along the C-111 canal, water, soil, plant tissue, and midge pupal exuviae samples were collected from sawgrass habitats at 4 pairs of sites. One site of each pair was located in sawgrass habitat approximately 100 m from the C111 canal, the other site was located about 1 km south (downstream) of the near-canal site.

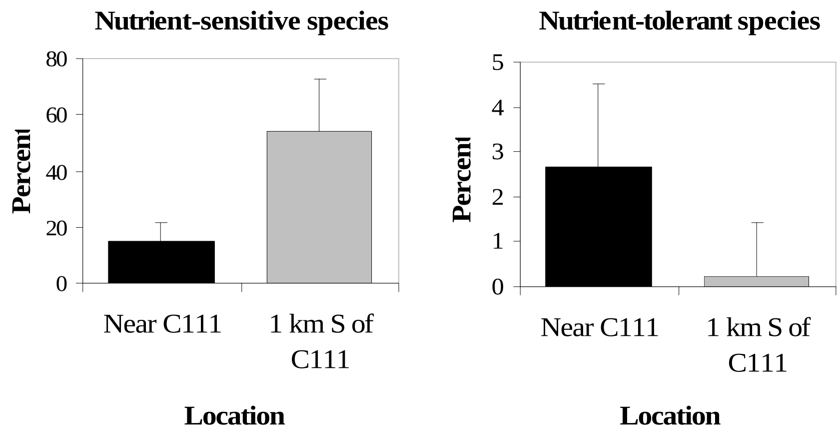


Figure 10. The relative abundances of nutrient-sensitive and nutrient-tolerant species in *Cladium* habitats close to, and approximately 1 km downstream from C-111 canal inflows to Taylor Slough, October 2002.

Changes in relative abundances of nutrient-sensitive and nutrient-tolerant species groups indicated that, in the eastern Panhandle of ENP, midge-community composition changed significantly near C-111 canal (Figure 10). Of the 16 species that either Jacobsen and Perry (2002) or King (2001) considered to be indicators of excellent water quality, 11 species were collected from Eastern Panhandle sites approximately 1 km distant from the C-111 canal, and 9 species were collected at sites near the canal. All but three species considered indicative of natural oligotrophic conditions declined in relative abundance near the C-111 canal. This response variability between species likely reflects variation in their sensitivity to enrichment. Species may eventually need to be ranked or scored according to their sensitivity to canal inflows and nutrient enrichment. Certain species such as *Cladotanytarsus* sp. A appear to be particularly sensitive to nutrient enrichment. Other species such as *Dasyhelea cincta* and *Tanytarsus* sp. D may actually benefit from low levels of nutrient enrichment. The lack of observed declines in *Dasyhelea cincta* suggest that, although nutrient levels appear to be higher near the canal, they were not high enough to cause breakup of calcareous periphyton mats. Eight of the 21 species that are indicative of nutrient enrichment based on studies by Jacobsen and Perry (2002) and King (2001) were collected in Eastern Panhandle samples. Six of these 8 species were collected only at sites near the C-111 canal.

2.2.4 Conclusions

In the Taylor Slough basin near the C-111 canal, shifts in proportions of nutrient–sensitive and nutrient-tolerant species groups suggested that marshes along the canal might have received a degree of nutrient enrichment from the canal.

Soil samples from detention areas at S-332B and S-332C indicate that these areas are enriched in comparison with adjacent marshes in the Rocky Glades (see Nutrients section of this report). Midge communities in all three detention areas are profoundly different than midge assemblages found elsewhere in ENP. Several species that were abundant in those detention areas (but otherwise rare in ENP) are well known to reach nuisance levels in shallow, enriched waters in tropical and subtropical regions of the world.

Midge communities showed strong shifts in relative abundance of indicator species in marshes near the S-332B and S-332C detention areas. It is possible that these community shifts represent a “spillover” effect from midge communities in the canal and detention areas. But this spill-over or “source-sink” hypothesis cannot explain the decrease in species that are sensitive to enrichment as one approaches the detention areas. Continued monitoring is needed to determine if the detention areas will be a troublesome source of nutrients in the future. Preliminary observations of midges in 2003 samples indicated that this pattern is persistent.

Comparisons of midge assemblages collected from 22 sites in eastern ENP in 1998-1999, and analyses of midges along the course of Taylor Slough in 2001, indicated that Taylor Slough marshes upstream of the Taylor Slough bridge, and particularly adjacent to the L-31W canal near S-332, have been degraded by polluted water and/or mismanagement of flows entering the Taylor Slough basin. The shift we observed in community composition toward greater representation of nutrient-tolerant species as one approaches the L-31W canal corroborates other studies of Taylor Slough invertebrate and periphyton communities (Newman et al. 2002). Our analyses also suggest that the conditions of marshes near S-332 have not improved since operational procedures changed in 2000, and water quality in the L-31W canal may still be a problem.

3. Hydrological Impacts

3.1 Rocky Glades

3.1.5 Invasion of non-native zooplankton through ground-water seepage: Effects on communities of possible structural changes along the eastern border of ENP.

Our baseline studies on surface- and ground-water microcrustaceans in the Rocky Glades of ENP and outside its eastern border have shown that the composition of ground-water communities is strongly determined by surface water communities and seasonal hydrological events (Bruno et al. 2003b). In ENP, surface microcrustaceans (planktonic copepods and cladocearns) colonized ground-water mainly at the beginning of the wet season, when rainfall recharged the aquifer, transporting microcrustaceans from permanent waters (solution holes, ponds, sloughs) into groundwater. Colonization also occurred at the end of the wet season, when ground water levels dropped about 60 cm below ground level and surface organisms follow the recession of the water table (Bruno and Perry, in press). Surface-water organisms entered ground water, but decreases in community similarity with increasing distance between wells suggested they did not disperse far from the input location. Along the canal system on the eastern border of ENP, ground-water communities were dominated by surface-dwelling copepods that dispersed into the aquifer following groundwater seepage directed eastward along canal L-31N (Bruno et al. 2003b). Higher numbers of copepods were collected in the dry season, when the lowering of water levels in L-31N canal increased groundwater flow eastward (Nemeth et al. 2000). Surface copepods may have entered the shallow aquifer, moving horizontally from canals. Several studies reported that the ratio of epigean to hypogean organisms in ground water reflects the amount of surface water influence and its seasonal variation (Danielopol et al. 1997, Gibert et al. 1995). When surface water infiltrates the aquifer, epigean organisms penetrate ground waters (Mösslacher 1998). Surface-water planktonic communities in the Rocky Glades during the wet season are largely determined by the re-emergence of dormant stages, and the dispersal of organisms from ground-water refugia to surface water after re-wetting. Both processes are related to hydrological factors such as hydroperiod and the extent of drydown (Loftus and Reid 2000; Bruno et al. 2001). The cyclical continuity between epigean and hypogean communities is strongly dependent upon the hydrological regime.

Surface populations of zooplanktonic copepods from water bodies north of ENP are easily transported into and carried through the canal system that borders ENP. Rainfall and seepage into ENP is the main source of surface water for the Rocky Glades and Taylor Slough, together with input from detention areas S-332B and S-332D, and from the degraded L-31W canal. However, studies on fish communities (Loftus, Kline, Trexler, this report) suggest the removal of connections with detention areas and canals as a way to prevent the introduction of more exotic fishes in ENP. For copepods, the ability to enter groundwater would be a critical step in colonizing surface waters in ENP from canals and detention ponds. As previously reported, most of the species collected in surface waters in ENP have shown the ability to colonize groundwater, and survive there for a length of time (Bruno et al. 2003a, b; Bruno and Perry, in press). The ability to colonize groundwater might represent a barrier, and thus a filter, for potential invasive species transported from permanent water bodies north of ENP. Once into ENP, the need to undergo diapause during the dry season would be a second mechanism limiting potentially invasive species from establishing permanent populations in ENP.

3.1.6 Drift Fence Study

It is difficult to assess the short-term effects of IOP in preserving and restoring natural ecologic and hydrologic features of the Rocky Glades because we have little information about the Rocky Glades before drainage affected the area. From historical accounts of the region (Willoughby 1898), and from backcast modeling of water levels prior to the Central and Southern Florida Project (Loftus et al. 1992, Renken et al. 2000), there is little doubt that both surface flooding and groundwater levels have declined. In all seasonal wetlands, aquatic species require a dry-season refuge to escape desiccation. In much of the Everglades, the historical refuge was the alligator hole (Nelson and Loftus 1996), but in the Rocky Glades, it was the solution hole. Loftus et al. (1992) and Kobza et al. (2004) discuss the ecological implications of groundwater reductions on the availability of dry-season refugia in solution holes for aquatic animals.

In May 2000, we established four drift-fence arrays to measure the dispersal and relative abundance of fishes with the arrival of the wet season. These were located on the Main Park Road, ENP, two west of the Pineland Trail, and two east of it (Arrays 1-4; Figure 11). The road shoulder served as a border to the south end of each array. We placed three minnow traps, with 3 mm square wire mesh and with one end of the trap blocked, at the center of each array to capture animals moving into the west, north, and eastern segments of each array. When the initial rise in the water table inundated the arrays, we soaked the minnow traps for 24-hour periods each day for two weeks. Later, as the water level stabilized, we reduced the frequency of sampling two trap-days per week for two subsequent weeks, then to one trap-day per week for the duration of the wetted period. In 2001, we established nine additional drift-fence arrays (Figure 11).

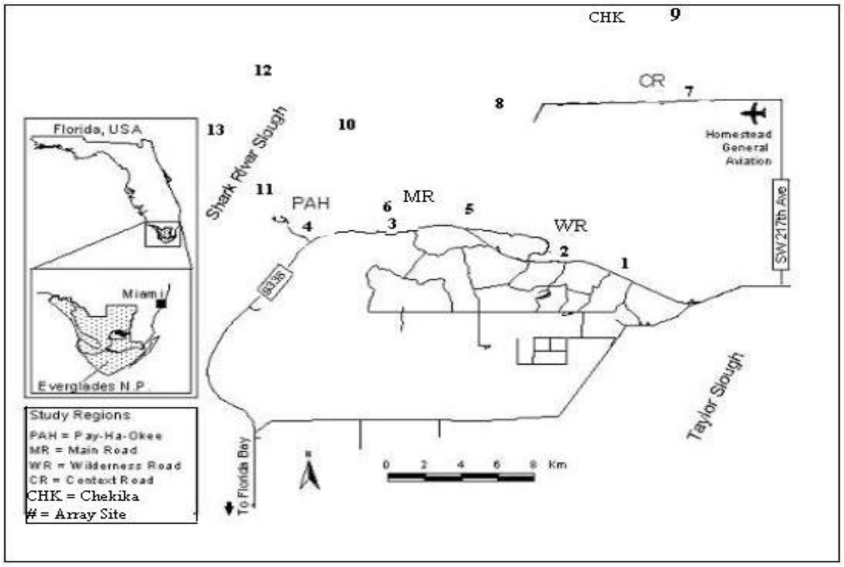


Figure 11. Location of drift array fences throughout the Rocky Glades

The typical pattern of fish catches at the arrays included a large pulse of activity at the beginning of the wet season, lower catches during the high water period of mid-summer and early fall, then another pulse of higher catches when water levels declined in fall and early winter (Figure 12). The initiation of fish catches on the wetland surface varied among sites and years. Array 4 had the longest hydroperiod and Array 1 the shortest. Arrays 7 and 9 had the very shortest hydroperiods of any site sampled. Water levels there were usually high enough for fish sampling only in late summer, when heavy rains flooded the area. In 2002, that situation did not occur and we were unable to collect samples there. Fishes in those areas are confined to solution holes for long periods and the populations are completely eliminated each year when the shallow holes dry. Until water levels are restored along the eastern boundary, it must be considered as a population sink for fishes.

Fish catches and relative abundances varied by year and by array. In general, arrays with the shortest hydroperiods had the lowest total catches. Hydroperiods at arrays 1, 2, and 3 range from short to moderate, but are shorter than those predicted by the Natural Systems model for the area. Array 4 with the longest hydroperiod, usually had the highest catches. Most sites appeared to show an inverse relationship between catch and water depth. This was most pronounced at Array 4. During high-water periods in 2001 and 2002, catches at maximum water levels declined at Array 4, probably because of dispersal into shallower sites and because of a trapping problem.

Figure 12. Catch per unit effort (CPUE) of fishes (bars) at Array 4 with water depth (line) in 2000, 2001, and 2002

Yearly rainfall patterns play a major role in determining the timing and duration of surface flooding, the number of reversals, and the extent to which groundwater levels fall in the spring. These patterns in turn affect the appearance and catch rates of fishes on the wetland surface, and probably also affect the interannual species composition. Our water-depth data from each array, and the data from ENP continuous recorders, demonstrate that surface and ground water conditions in the Rocky Glades are dependent upon and are highly responsive to rainfall. In addition to long-term reductions in water levels in the Rocky Glades, the data set for the easternmost arrays suggests that pumping and drainage operations during IOP affected hydropatterns and fish survival in those arrays. In 2002, average dry-season rainfall kept the below-ground recession rate at a moderate level until pumping from L-31W ceased (S. Mitchell, pers. comm.). This dramatically increased the rate of recession, drying the solution-hole refuges, and causing fish mortality. Similarly, pre-storm drainage actions in September 2002 dried the wetlands earlier than normal, forcing fishes into solution holes where mortality occurred over time. Conversely, at Array 4, flooding in 2002 extended into 2003 at this site. We believe that hydrology at this site is responding to operations that affected Shark Slough rather than the Rocky Glades/Taylor Slough area. Overall, hydrological conditions in the Rocky Glades, and subsequent impacts on the fishes, did not improve from the pre-IOP to the post-IOP period (Ahn 2003). Therefore, we conclude that IOP failed to provide the increased water depths and hydroperiods anticipated for this region and its aquatic biota.

3.1.7 Introduced Fishes

At the beginning of 2000, seven species of introduced fishes bred in ENP, and two species (including the butterfly peacock bass, *Cichla ocellaris*) had not yet established (Loftus 2000). Loftus (2000) noted that several species of exotic fishes occurred in canals bordering ENP but had not yet been observed or collected in the park. Prior to the summer of 2000, no new exotic species had been observed within ENP for several years (Kline 2000, Trexler et al. 2000, Kobza et al. 2004). Since the implementation of ISOP and IOP, we have collected three new non-native species in ENP and observed the peacock bass at a new area of ENP (Table 2).

Table 2. Non-native fish species first noted in Everglades National Park after opening of S-332 structures and lowering berm on L-31W.

Common Name	Species Name	First observed in ENP	Notes
jaguar cichlid	*Cichlasoma managuense*	Aug 2000	First appeared in the canal system in Dade Co. in 1990's
jewel cichlid	*Hemichromis letourneauxi*	Aug 2000	Very abundant in 2003
armored catfish	*Hoplosternum littorale*	Aug 2002	Invaded Dade Co. via canals from east central Florida around 2000
butterfly peacock bass	*Cichla ocellaris*	Nov 2002	Collected in L31W prior to 2002

With the implementation of ISOP beginning in 2000, water levels in the L31W canal were raised and overflowed over the bank to introduce sheetflow into upper Taylor Slough (Figure 13). Two species new to the Park were collected for this first time in August 2000, in areas near the S-332 structures. By October 2002, the biomass of non-native fishes was significantly greater in the vicinity of the S-332 detention areas than at locations further inside the Park (Figure 14).

Figure 13. Inflow point of water from the L-31W canal into Everglades National Park. The old S-332 pump is seen on the left and water now flows freely into the Park, on the top of the figure. This is the area where non-native fishes are thought to be entering ENP.

ISOP/IOP management actions may have directly contributed to the introduction of or the redistribution of new non-native fishes in ENP. It is likely that jaguar cichlids, armored catfish, and butterfly peacock bass entered Taylor Slough from the L-31W canal because of ISOP or IOP management actions. Although the jewel cichlid was first collected north of the L-31W canal, they presently inhabit the retention areas and have dispersed far into the Rocky Glades and C-111 areas, possibly as a result of ISOP/IOP actions. The connectivity of the canal system and redistribution of water may have helped to promote the range expansion of these species throughout south Florida, because canals act as dispersal corridors and refuges for introduced species (Howard et al. 1994, Trexler et al. 2000). Loftus et al. (2003) listed as many as 10 additional non-native fish species residing in the eastern canal system that are potential future invaders of ENP with the help of water-management structures and operations. Exotic aquatic organisms must be a significant consideration in planning future water deliveries into ENP.

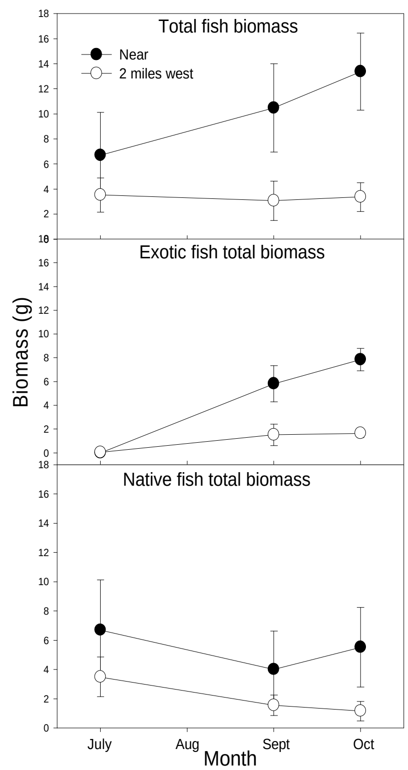


Figure 14. Biomass (g) of all fish, exotic fish, and native fish collected by minnow traps in Everglades National Park marshes near (solid symbols) and 2 miles west of the S-332 detention areas in July and October 2002. Data are mean +/- 1 S.D.

Most agencies agree that control and management of non-indigenous species is a critical component of ecosystem restoration in South Florida (2003 Everglades Consolidated Report). Control and management of invasive exotics are priorities established by the South Florida Ecosystem Restoration Task Force (SFERTF) in 1993. The first annual report of the South Florida Ecosystem Restoration Working Group (SFERWG) in 1994 addressed all non-indigenous plant and animal species in the region calling for halting or reversing the spread of invasive exotics, eradicating where species are contained, and preventing the introduction of new invasive species. The U.S. Department of the Interior's (DOI's) Fish and Wildlife Coordination Act Report (FGFWFC, 1999) for the CERP also considers control and management of non-indigenous species to be a critical aspect of ecosystem restoration in south Florida. This should be a major consideration in planning structural and operational changes in IOP.

3.1.8 Stranding event

In February 2000, early in the ISOP period, water inflow was stopped abruptly to reduce water levels where Cape Sable Seaside Sparrows nest. This drastic decrease in water levels due to water management caused a stranding fish kill event in upper Taylor Slough (Kline 2000). Prior to the halting of inflows, water levels in upper Taylor Slough were being maintained above ground surface by pumping at the S-332 pump station. A decrease in discharge beginning on February 18^{th} caused water levels in the upper region of Taylor Slough to decrease below ground rapidly. Many groups of stranded fishes were observed on 23-24 February 2000 in an approximately 1ha area about 300 m north of pump S-332. Twelve species of fishes composed an average density of 183 ± 81 m^2 and biomass of 97 ± 30 g/m^2 (n=5) within the area of stranding. Eight orders of macroinvertebrates were also found stranded with the fishes. There was evidence of crayfish (*Procambarus alleni*) burrowing, however many crayfishes were seen moving lethargically and drying on top of matted vegetation and on the substrate. During the collections on February 23-24, no wading birds were observed feeding in the dried areas of the marsh.

3.2 Wet Prairie/Sloughs

3.2.1 Vegetation

We monitor the floating mat in wet-prairie sloughs at 22 locations in Shark River Slough, Taylor Slough, and WCA-3A and WCA-3B. The floating mat volume is a good indicator of the mass of periphyton in a marsh (Periphyton ash-free dry mass (g) = -10.2 + [0.022 X floating-mat volume (ml)]; $R^2 = 0.77$, $P < 0.001$, n = 16 Turner et al. 1999). In a recent study we tested for visitor impacts on plant and aquatic animal characteristics at our long-term study plots but found no evidence for them (Wolski et al., in review).

The volume of floating mat decreased in the IOP/ISOP period relative to the pre-ISOP/IOP period at all but three study sites, though the change was significant only at four sites in northern Shark River Slough, one site north of I-75 in WCA-3A, and a site in northwest WCA-3A south of I-75 (Table 3). All of the sites with significant change before and after IOP displayed volume decreases of 70% or more. Three sites yielded estimated changes by more than 60% after the change of operations, but could not be judged as significant because of high variability among plots and years within the before and after IOP/ISOP time periods.

The general trend of decline in mat volume is contrasted with a regional pattern in stem density of emergent vascular plants (Table 3). Stem density declined at 9 of 11 sites in Shark and Taylor Sloughs, but increased at 9 of the 11 sites in WCA-3A and WCA-3B. There were no regionally consistent temporal changes in the relative abundance of spikerush at these study sites (Table 3), but it comprised 70% on average of the emergent stems at these sites. Thus, the regional patterns of change in emergent vascular plant stem density were not the result of changes in dominance of the most abundant plant at each site.

Table 3. Summary of BACI tests for effects of IOP on aquatic plants in WCA-3A, WCA-3B, Shark River Slough, and Taylor Slough. The percentage change from pre-IOP to post-IOP is reported. Changes indicated as significant (P<0.05) by a Tukey's post-hoc test are indicated in colored boxes. Emergent stems is the total number of vascular plant stems (number/m^2).

Region	Study Site	Floating mat	Emergent stems	spikerush relative abundance
SRS	6	-73.9	-33.4	3.3
SRS	23	-81.5	-31.0	-10.7
SRS	50	-86.7	-71.4	-22.8
SRS	7	-80.3	-14.7	-8.7
SRS	8	-19.8	-21.9	-0.2
SRS	37	-64.2	26.7	0.6
Taylor	MD	-68.2	-1.7	-1.4
Taylor	MDsh	-24.2	-51.6	-43.1
Taylor	TS	20.9	-11.4	-8.7
Taylor	TSsh	-41.4	-30.8	-7.9
Taylor	CP	-7.0	-63.1	-14.4
WCA	1	-28.1	-41.6	-48.9
WCA	2	-17.3	187.7	-2.4
WCA (western 3A)	3	73.2	69.9	1.2
WCA	4	-94.1	112.6	-2.1
WCA	5	-56.0	159.6	-2.4
WCA	6	-14.9	2.7	-0.4
WCA-3B	7	-46.0	83.0	11.6
WCA-3B	8	15.6	26.7	-0.2
WCA - north of Alley	9	-92.2	54.4	13.9
WCA- north of Alley	10	-52.7	54.0	18.2
WCA (western 3A)	11	-76.4	-17.6	-41.6

KEY
increase
decrease

3.2.2 Small fish and grass shrimp

We monitored small fishes (<8-cm standard length) and macroinvertebrates throughout the pre-IOP and IOP/ISOP period in wet-prairie sloughs in Taylor Slough, Shark River Slough, and WCA-3A and 3B (Figure 15. work in progress). The fish were collected in 1-m^2 throw traps to provide estimates of density. Through analysis of taxa demonstrating monotonic responses to time since marsh drying (Figure 16), these data permit an assessment of hydrological effects of the IOP. We emphasize five taxa chosen as Performance Measures for assessing hydrological influence on aquatic communities in wet prairie habitats. Three fishes (bluefin killifish, least killifish, and flagfish) and one macroinvertebrate (riverine grass shrimp) were chosen because of their sensitivity to drying events (all but flagfish decrease from high mortality they incur). We also evaluated IOP/ISOP impacts on a third fish species because of its important role in the Everglades food webs (eastern mosquitofish). Finally, we report an analysis of fish density (all fish species summed), as an overall indication of operations on fish availability to higher trophic levels. We observed identical results when analyzing these data on fish density (number/m^2) or biomass (g/m^2) so we report only density.

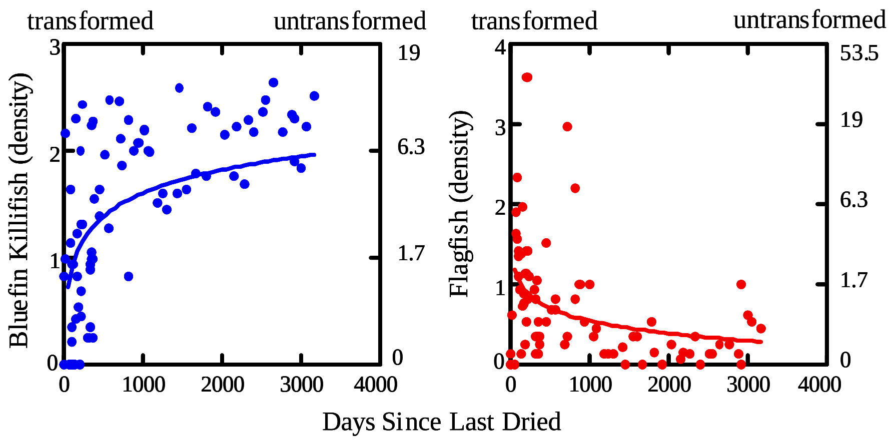


Figure 16. Change in density of two species of small fish used as performance measures for hydrological assessments. The left axes indicate transformed scale used for statistical analyses (ln (#/m^2 + 1)) and the right axes indicates untransformed density.

In all cases our tests of the BACI hypothesis indicated significant change before and after the IOP/ISOP period that varied among sites within regions (i.e., the difference in AIC values between models with and without the site-by-treatment interaction exceeded 2). Aquatic animals known or suspected to incur high mortality and recover primarily by local reproduction (rather than long-distance dispersal) displayed lower densities in the IOP/ISOP period than in the preceding four years (Table 4). These species are bluefin killifish, least killifish, flagfish, and grass shrimp. The density of those taxa decreased by 50% or more at most study sites in Shark River Slough, Taylor Slough, WCA-3A north of Alligator Alley and WCA-3B; flagfish density increased at these sites (Figure 17). Of the three study regions (WCA-3, Shark River Slough, and Taylor Slough), only Taylor Slough experienced widespread marsh drying in the pre-IOP/ISOP years (four of the five study sites there dried in 1999). The pattern among sites was mixed in WCA-3A south of Alligator Alley, consistent with known patterns of marsh drying there (Figure 18). Sites WCA3 and WCA11 are located in western WCA-3 and experienced marked drying events in 2000. These sites displayed patterns similar to those in Shark River Slough. However, sites WCA1, 2, 4, 5 are located in central or eastern WCA-3A and dried less severely or not at all; density of small fish actually increased in the IOP/ISOP period at these sites.

Eastern mosquitofish are among the most abundant fish in the Everglades (Loftus and Kushlan 1987, Trexler et al. 2002) and are aggressive predators on juvenile fishes and macroinvertebrates (Taylor and Trexler 2001; Geddes and Trexler 2003) that are important links in the Everglades food web. They are consistently among the first fish to re-colonize a habitat after a drying event and are considered strong dispersers (Trexler et al. 2003). The impact of IOP/ISOP on their density patterns was heterogeneous among sites and was statistically significant at only five of 22 studied. However, their density increased at WCA1, 2, and 4, probably contributing to the increase in total fish density at sites WCA1 and 4.

Table 4. Summary of BACI tests for effects of IOP on aquatic communities in WCA-3A, WCA-3B, Shark River Slough, and Taylor Slough. The percentage change from pre-IOP to post-IOP is reported. Changes indicated as significant (P<0.05) by a Tukey's post-hoc test are indicated in colored boxes.

Region	Study Site	Total Fish	bluefin killifish	least killifish	grass shrimp	flagfish	eastern mosquitofish
SRS	6	-7.1	-36.5	-9.8	21.0	31.7	46.9
SRS	23	-16.5	-28.0	-9.3	16.3	-1.1	10.6
SRS	50	-59.8	-45.0	-27.7	-17.8	10.6	-44.9
SRS	7	-50.6	-71.2	-77.2	-50.8	108.6	-6.2
SRS	8	-41.4	-60.9	-63.2	-56.7	41.1	-16.4
SRS	37	-37.3	-58.9	-66.0	-66.9	38.5	-4.9
Taylor	MD	-59.8	-59.9	-75.4	-42.5	13.3	-22.6
Taylor	MDsh	-65.4	-41.9	-53.3	-39.1	-28.6	-24.5
Taylor	TS	-54.8	-57.3	-60.4	-59.4	-13.5	-26.1
Taylor	TSsh	-37.6	-35.5	-35.5	-39.0	-8.2	-10.7
Taylor	CP	-26.8	-57.1	-15.8	-35.9	39.8	-3.2
WCA	1	74.8	26.0	74.3	68.2	63.9	125.2
WCA	2	29.2	12.7	22.7	23.6	94.1	55.9
WCA (western 3A)	3	-23.9	-44.6	-38.7	-65.2	52.0	-10.3
WCA	4	71.3	22.1	70.7	52.9	22.8	74.8
WCA	5	27.0	28.0	13.3	-12.4	77.2	5.5
WCA	6	53.9	19.9	42.9	-3.0	36.0	22.6
WCA-3B	7	-31.1	-38.4	-48.1	-41.8	45.6	-29.2
WCA-3B	8	-27.0	-44.6	-54.6	-51.5	55.2	-3.1
WCA - north of Alley	9	-44.4	-38.1	-47.4	-63.5	32.6	-34.2
WCA- north of Alley	10	-33.1	-18.0	-41.2	-53.5	84.3	-28.2
WCA (western 3A)	11	-42.7	-60.8	-44.4	-86.3	-52.8	-26.2

KEY
increase
decrease

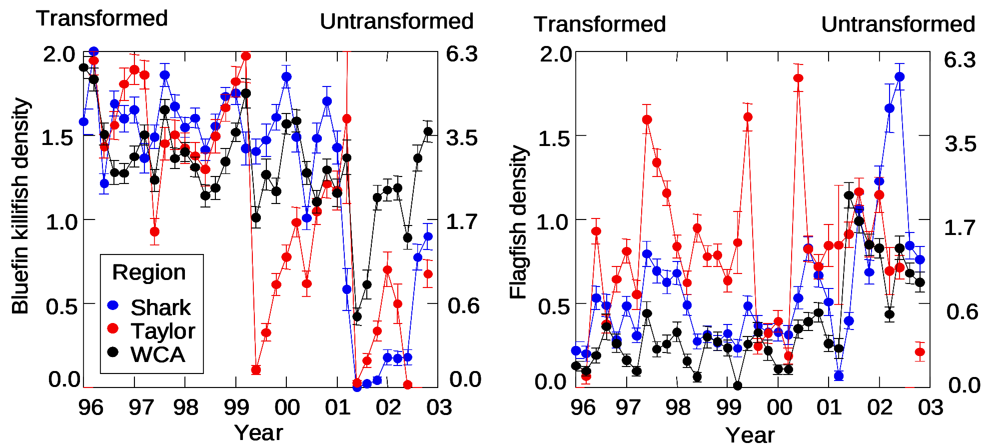


Figure 17. Density of two species of fish illustrating time series patterns. Transformed density (ln (#/m^2 + 1)) was used to meet assumptions of the analyses, untransformed density is #/m^2. Mean +/- one standard error are plotted.

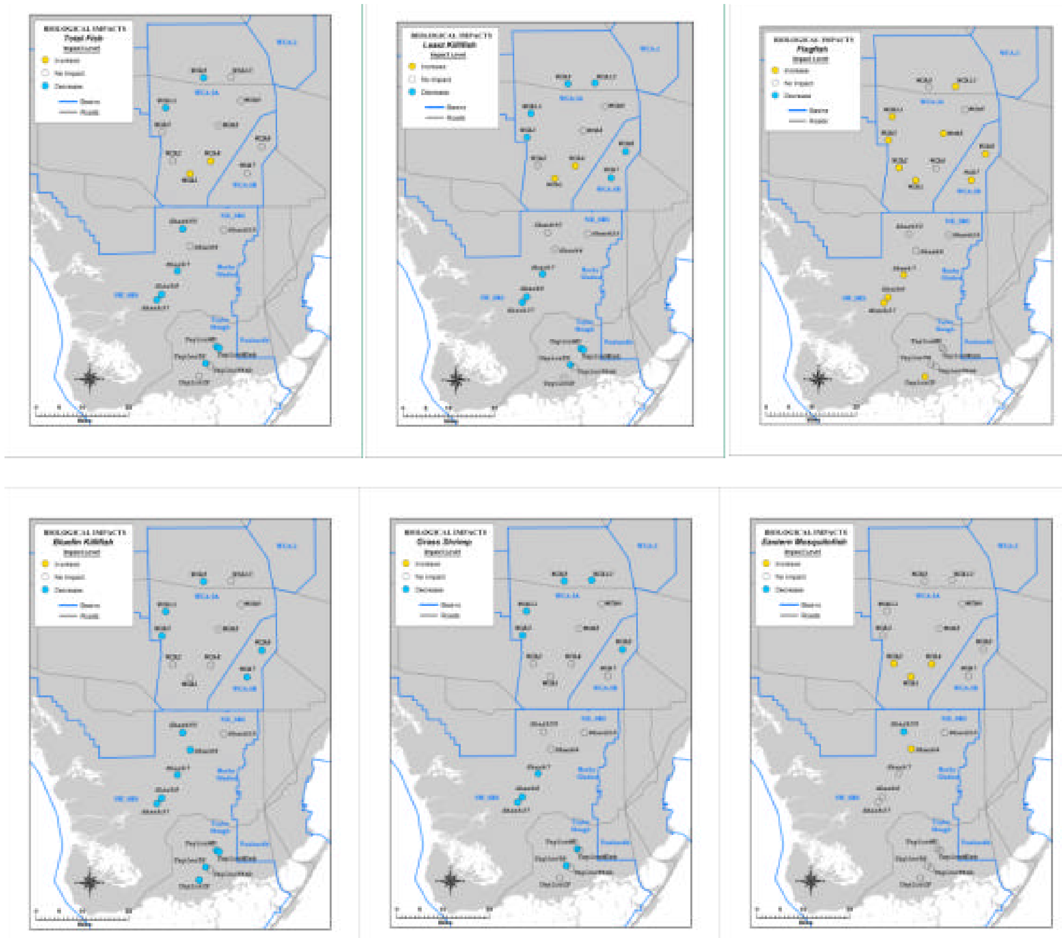


Figure 18. Maps illustrating the spatial distribution of change in aquatic animal density from before and after IOP/ISOP

3.2.3 Large Fishes

We monitor large fishes (standard length > 8 cm) using an electrofishing unit mounted on our airboat (Chick et al. 1999). Data were collected by electrofishing 5-minute transects (9 per study site per visit). Each transect was separated by at least 50-m and covered approximately 150 to 250 m of marsh with the airboat running at idle speed (4 - 8 km/h). All electrofishing was conducted between 0730 and 1700 hours (Chick et al., unpublished manuscript) at one of 11 study sites (Figure 19). We sample four times per year (February, April, July, and October) and sampling began in 1997 and continues to the present. We analyzed the CPUE of all species summed and the CPUE of piscivores (largemouth bass, Florida gar, warmouth, yellow bullheads, and chain pickerel).

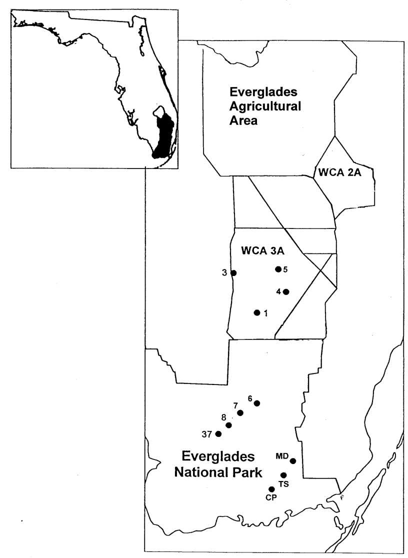


Figure 19. Map of sites used for monitoring big fish by electrofishing. All sites are also sampled by throw trap.

Analysis of both total CPUE and catch of predatory fish before and after the IOP/ISOP period yielded significant changes that differed among the study regions (the difference in AIC values between models with and without the region-by-treatment interaction exceeded 2). In this case, the changes matched our *a priori* hypothesis that the magnitude of effect was greatest in WCA-3A, and intermediate in Shark Slough (Table 5). The CPUE of all fishes decreased by over 25% in both Shark Slough and WCA-3A in the IOP/ISOP period compared to the three years prior, with no significant change in Taylor Slough. The CPUE of piscivores at our marsh study sites dropped significantly in WCA-3A under IOP/ISOP.

Table 5. Summary of BACI tests for effects of IOP on big fish in WCA-3A, WCA-3B, Shark River Slough, and Taylor Slough. Log- transformed mean CPUE are reported. Diff is the percentage from pre-IOP to post-IOP reported on scale of raw data. Changes indicated as significant when P<.05 as determined by a Tukey's post-hoc test.

Variable	Region	1996-1999 before	2000-2002 after	diff	P-adj
Predator	Shark River Slough	.4384	.3598	-7.6	.9007
	Taylor Slough	.2985	.1965	-9.7	.8061
	WCA-3A	.7891	.4915	-25.7	.0009
CPUE	Shark River Slough	1.2469	.8174	-34.9	.0002
	Taylor Slough	.6005	.4952	-10.0	.9192
	WCA-3A				

Taylor Slough, Shark Slough, and WCA-3A had distinctly different large-fish communities in the 1997-1999 period prior to IOP/ISOP (Figure 20A). The IOP/ISOP period, starting with the dry-downs of 2000 in WCA-3A, caused the WCA-3A and Shark Slough communities to become more similar, converging on a species composition similar to that of Taylor Slough in the pre-IOP/ISOP period (Figures 20B; 21). WCA-3A had a much more abundant piscivore community (particularly largemouth bass) prior to 2000. This part of the community appears to have suffered high mortality in the dry-downs of 2000. This led to pervasive changes in the fish community of WCA-3A. This is illustrated by a change in the size distribution of lake chubsuckers in WCA-

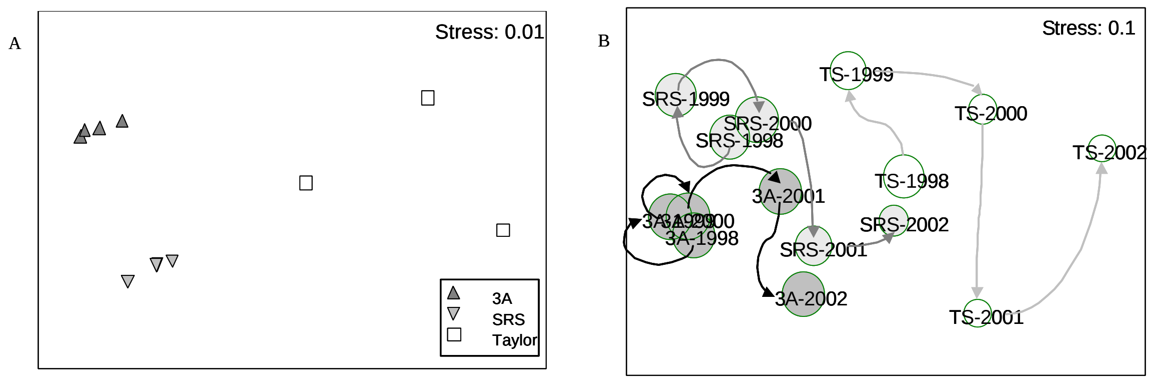


Figure 20. A. Two-dimensional non-metric multi-dimensional scaling (NMDS) ordination of community structure. 3A = WCA-3A, SRS = Shark River Slough, TS = Taylor Slough. B. Two-dimensional NMDS ordination of regional community structure through time. Circle size is scaled to the natural log of the number of days since the last dry-down event (i.e., since mean water depth < 10 cm). Mean CPUE data were square-root transformed and Bray-Curtis similarities were used for both ordinations.

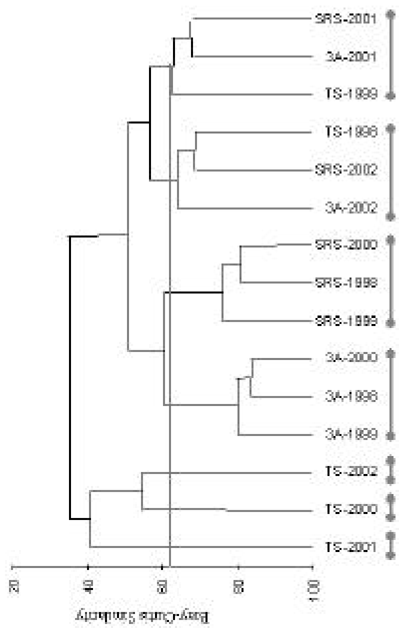


Figure 21. Cluster analysis of Bray-Curtis similarity indices illustrating community change in three study regions from 1998 to 2002. Note that 1998 to 2000 clustered together for SRS and WCA-3A, but separately from the later years.

3A. Prior to 2000, WCA-3A had a larger size (and older) distribution of this species than in Shark Slough or Taylor Slough (Figure 22). Following 2000, the size distribution (and therefore age structure) in WCA-3A shifted to match that of Shark River Slough.

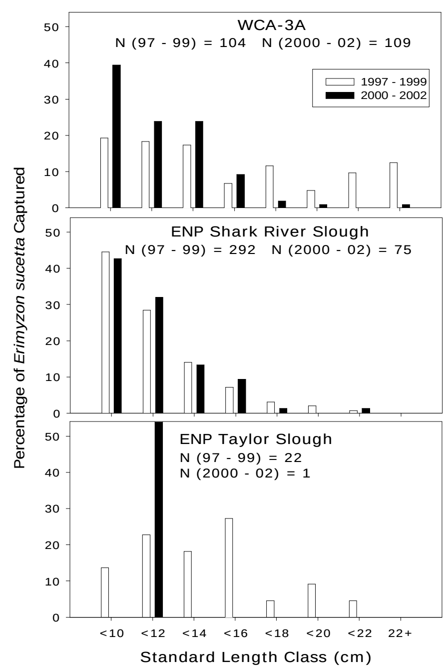


Figure 22. Length-frequency distribution of lake chubsucker (*Erimyzon sucetta*) captured in WCA-3A, Shark River Slough and Taylor Slough from October 1997 to October 1999 (white bars) and from February 2000 to October 2002 (black bars).

The widespread nature of changes in hydrological management associated with IOP/ISOP and the coincidence of the pre-IOP/ISOP period with relatively wet years and IOP/ISOP with drier ones, limits our ability to make strong statement about its impacts to date. However, it is clear that the IOP/ISOP period corresponded with profound changes in the community structure and function of food webs in wet prairies of WCA-3 and ENP. The drying in WCA-3A that was greater than expected based on rainfall diminished the abundance of large piscivorous fishes there, led to changes in the abundance of small fishes and aquatic invertebrates, and certainly altered trophic dynamics there for several years. Wading bird nesting was particularly strong in 2002 (Gawlik 2002), possibly linked to the changes in they prey community that tend to unfold one or more years after drying events (Frederick and Ogden 2001).

4. Summary of Impacts

Nutrients

- Elevated periphyton TP at S-332B, possibly at S-332D
  - Enrichment limited to immediate vicinity of input
  - S-332B enrichment may result from groundwater seepage

- Retention ponds harbor variety of species atypical of nearby Rocky Glades/marl prairie habitats (include taxa indicative of elevated nutrients and "ponded" habitats).

- Midge-indicator taxa provide evidence of nutrient impacts by disappearance of taxa intolerant of enrichment, and increases in taxa associated with enrichment.

Hydrology

- Floating mat volume decreased following marsh drying events in Shark River Slough and WCA-3A

- Copepods atypical of marl prairies were observed in ENP inside retention ponds S-332B and S-332D

- Grass shrimp abundance declined at all sites where marshes dried. Recovery is slow (requires 1+ years of inundation)

- IOP construction/operation facilitated spread of non-native species into ENP marshes and range expansion of native species atypical for habitats
  - *New S-332 structures provide direct access to ENP marshes from L-31W. Completion coincides with first collections of three new species of non-native fishes and range expansion of a fourth, plus range expansion of two native fish species, into nearby Rocky Glades and Taylor Slough habitats.*
  - *Since spoil mounds south of C-111 were removed, two new non-native fish species observed in marl prairies and mangroves downstream.*
  - *Midge species typical of canals were observed in ENP wetlands near S-332B, S-332C, and S-332D overflow, and near C-111 in panhandle region.*
  - *Retention ponds and canals have high diversity and density of exotic fish, midge, and copepod taxa and are potential sources for primary and secondary invasions.*

- Annual minimum groundwater depths in Rocky Glades area were too low in Test 7 and IOP failed to improve this.
    - *Current minimum water levels leave few solution holes inundated to act as aquatic refuge through the dry season*
    - *Pre-storm drawdowns and other operations appeared to increase recession rates of water and resulted in immediate fish mortalities.*
    - *Reversals in Rocky Glades may diminish the spawning stock of aquatic species that survive the dry season*
    - *Each re-flooding event compounds losses of fishes and diminishes over-winter survival.*

- IOP lowered water levels in the dry season below levels expected from Test 7 and led to lower densities of hydrologically sensitive species
    - *Observed lower density and relative abundance of large predatory fishes in WCA-3A, SRS, and Taylor Slough coincidentally with IOP/ISOP (largemouth bass and gar). Change in age structure of lake chubsuckers (shift to younger age structure in WCA-3A) occurred at same time.*
    - Fewer small fishes in western 3A and in 3B, with MORE of some species in southeastern 3A were noted. The increase may have resulted from predation release in longer hydroperiod areas following large-fish mortality.
    - Our data suggest pervasive food-web effect in WCA-3A, with effects on the canal-associated fishery (L-67 and Tamiami) in 2001, 2002.
    -

5. Future Monitoring Needs: Aquatic Communities

We suggest that the goal of biological monitoring program should be to construct an integrated baseline program that is maintained consistently through time for impact assessment. Monitoring should focus on nutrient and hydrological impacts, with additional efforts for toxics, particularly mercury. Also, the CESI program (or a subsequent program) should continue to fund targeted research initiatives to improve interpretation of monitoring data.

Budget requests are listed on an annual basis for 2004. A 3% increase should be added annually to account for cost increases.

5.1 Baseline Monitoring Program

5.1.4 Wet-prairie sloughs

Throw-trap/electrofishing study: This project currently monitors small and large fish and large macroinvertebrates in wet prairies of Shark River Slough, Taylor Slough, and WCA-3A and B. Throw trap and airboat-mounted electrofishing are used. It focuses on gathering data for time-series analyses (currently 8 years at 20 sites; 25 years at three of these) that can be used to track trends of forage species (food for wading birds and other predators) over time. Fish in 34 alligator ponds are also monitored four times per year by electrofishing. Recent work has demonstrated that throw-trap samples gathered by this study are an excellent tool for monitoring crayfish. Crayfish data are not reported here because pre-2000 identifications of crayfish are currently being re-evaluated; a crayfish species new to the Everglades was identified after this work was begun so prior data have to be re-visited. The project also tracks game fishes with airboat-mounted electrofishing unit, and invasive fish species with both techniques. The throw-trap technique has been chosen for use in CERP-MAP, and these data will integrate with that effort. $225,000 per year

5.1.5 Rocky Glades, including Taylor Slough Headwaters

Drift-fence study: These habitats are not amenable to the techniques used in wet prairies because of the irregular surface topography and difficulty of access (no established airboat trails and often too rocky and shallow; access to sites not adjacent to roads requires helicopter). This project provides time series of aquatic-animal data that track colonization, succession, and exotic incursions with changing water depths, and other useful measures. Funding for drift-fences along the Main Park Road ends in FY04. Work in other areas of Rocky Glades and Taylor Slough Headwaters (near S-332 structures) is funded through FY05. A current effort seeks to relate the data collected by this technique to data from throw traps in sloughs to standardize results across the landscape. This project also monitors 25 solution holes after water recedes below the marsh surface to record the effects of groundwater levels on animal survival and on the success of exotic fishes. $150K per year for Park road and solution holes & $80,000 per year for Taylor Slough headwaters/S-332

Midge indicator species: This technique has proven very powerful for tracking nutrient enrichment in the Everglades and incursions of species atypical of wetlands from canals and detention areas. Current work in the areas downstream from the S-332 structures and the C-111 canal needs to be expanded and continued. Also, routine sampling down gradient from the S-12 structures and inflows into NE Shark River Slough is needed. $120,000 per year

5.1.6 Detention Areas/C-111 basin/Panhandle Area

Currently there is only sporadic sampling in these areas, but they are important for tracking the potential and actual invaders of park habitats (swamp eels and other non-natives recently discovered in C-111, but not yet seen in Park). The throw-trap study should expand to include four sites (close to C-111 and downstream, near mangrove fringe) and include electrofishing for large gamefish. It should include minnow trapping/drift fences in detention ponds and park inflow areas along C-111. $65,000 per year

5.1.7 Periphyton

A network of 800 monitoring sites has been established in the marl prairie, primarily in areas designated as habitat for the Cape Sable Seaside Sparrow (populations A-H). Approximately 300 sites are visited each year during the CSSP nesting season, at which time periphyton characteristics are recorded and an extensive vegetation survey is performed. Half of the sites are re-visited during the following wet season to collect periphyton, which is then processed for biomass and compositional analysis. Periphyton attributes will be mapped and related to hydroperiod and vegetation characteristics. $90,000 per year

5.2 Sporadic Sampling

Mercury in fishes as indicators: An analysis of the distribution of mercury in mosquitofish and largemouth bass should be conducted once every five years to assess status of this toxic compound in the environment. EPA-REMAP has started such an initiative, which should be fostered. This also provides water, soil, and floc nutrient analyses across the Park, which proved important for recent nutrient assessments.

5.3 Research Supporting Monitoring

a. Continue development of midge-indicator taxa. $100,000 per year – 2 years

b. Continue development of techniques for monitoring in Rocky Glades habitat. $60,000 per year – 2 years

c. Food-web effects of nutrient enrichment in short-hydroperiod communities. $130,000 per year – 2 years

d. Continue experimental studies of interaction of drying on periphyton communities to improve interpretation as nutrient indicator group in short-hydroperiod habitats. $60,000 per year – 2 years

5.4 Other Data Needs

Better topographic information is necessary for doing predictive hydrologic and biologic work in the Rocky Glades. Even the recent high-resolution topography is not adequate because measurements were taken every 400m along transects in an area where topography varies greatly on much finer scales.

6. Recommendations (Structural and Operations)

The authors affiliated with Florida International University and Everglades National Park have developed the following recommendations for operations and structural alterations of water management to ENP. These address three areas are consistent both with CERP goals and those of the Park: "Get the water right" in Northeastern Shark River Slough, the Rocky Glades, and Taylor Slough; exclude further damage to ENP wetlands from nutrient enrichment; and limit further expansion of non-native aquatic fauna into ENP property.

Figure 23. Aerial view looking west over agricultural lands on eastern border of ENP. The S-332D structure is visible. "Marsh ops" seek to minimize seepage of nutrient-enriched water from lands adjacent to Park's boundary into its wetlands.

- Successfully implement "marsh ops" concept to decrease seepage from ENP toward eastern boundary lands (Figure 23)

- Monitor nutrient inflows at S-332B, C and D and inside the detention areas
  - *Motivation: Preliminary evidence is consistent with nutrient impacts from introduction of water from B and from inflow water to ENP from D*
  - *Observed nutrient spikes in S-332B that might move into groundwater and subsequently into ENP*

- Eliminate high head cell in D while providing conveyance of water from S-332D west
  - *Motivation: Reduce seepage losses to C-111 canal*

- Fill in L31W or otherwise isolate canal and retention ponds from ENP marshes.
  - *Motivation: canals and retention ponds harbor non-native taxa and appear to be expediting invasion into ENP*

- Increase water levels in SS and NESS
  - *Motivation: IOP has created dry conditions when compared to Pre-IOP; have not moved toward long-term CERP goals of increasing hydroperiod in these areas*

- Manage Rocky Glades and Northern Taylor Slough through introduction of water further north and controlling seepage through C-111 project features

- Maintain storage in WCA3A through elimination of Zone E1
  - *Motivation: Elevated P in WCA3A and SS inflows*
  - *Use S-151 and move excessive water from WCA3A to WCA3B (accelerate MWD features in L67A/C)*

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